generally have an effect on supply of output and on the demand for inputs. The introduction of a new variety would affect the demand for agricultural chemicals. This change in the use of agricultural chemicals would provide the link from the economic behavior of the farmers to the physical and biological models used to quantify pollution externalities.

The production model also shows that, generally, the economic relationships in period t depend on the resource stocks and living organisms represented by R_t and S_t. The economic model does not determine these variables in the current production period, rather R_t and S_t play the role of constraints on the production process. The values of R_{t+1} and S_{t+1} in the next period are determined in part by the production decisions in period t. Thus the physical, biological, and economic sectors of the model interact dynamically according to the particular structure and parameterization of the systems of equations used to represent them. Given estimates of the parameters of these equations, initial values of the stocks R_t and S_t, and predictions of the "forcing variables" such as prices that are determined outside of the model, the system of equations can be used to generate predictions of the time paths of agricultural production (Q_t), input use (X_t), and the physical and biological stocks $(R_t$ and $S_t)$.

3.2 Long-run dynamic investment models.

In some cases it is not appropriate to use a short-run static production model to analyze externality generation. A long-run model may be needed for a variety of reasons: because the choice of capital stock is important in the amount of externality created; or because farmers do take externalities into account in their decision making; or for long-run regional analysis of externality creation where the effect of the externality feeds back into the production process. To illustrate, consider a modal in which physical capital evolves over time according to

$$Z_{t+1} = (1 - \delta)\, Z_t + V_t \,,$$

where δ is the rate of capital depreciation and V_t is the rate of gross investment each period. Similarly assume that the dynamics of the resources R_t and species S_t are given by

$$R_{t+1} = H(R_t, X_t, Z_t)$$

$$S_{t+1} = B(S_t, X_t, Z_t, R_t)\,.$$

The long-run maximization problem of the farmer is now defined as choosing the sequence of investments to maximize the present discounted value of profit from each period over the relevant planning horizon:

$$\underset{\{V_t\}}{\text{Max}} \sum_{t=1}^{T} \eta_t \left\{ \pi\left[P_t, W_t, Z_t, R_t, S_t, \tau_t\right] - U_t V_t \right\} + J\left[Z_{T+1}, R_{T+1}, S_{T+1}\right].$$

subject to:

$$Z_{t+1} = (1-\delta)Z_t + V_t$$

$$S_{t+1} = B(S_t, X_t, Z_t, R_t)$$

$$R_{t+1} = H(R_t, X_t, Z_t)$$

where η_t is a discount factor depending on the rate of interest, U_t is the price of investment goods, and J measures the terminal value of the physical capital and resource stocks.

The above problem can be solved using optimal control or dynamic programming techniques. For example, the solution can be obtained by maximizing the following Hamiltonian equation:

$$H_t = \eta_t \left\{ \pi\left[P_t, Q_t, Z_t, R_t, S_t, \tau_t\right] - U_t V_t \right\} + \lambda_t \left\{ (1-\delta)Z_t + V_t \right\}$$

$$+ \mu_t B(S_t, X_t, Z_t, R_t) + \rho_t H(R_t, X_t, Z_t)\,,$$

where λ_t, μ_t, and ρ_t are the multipliers for Z_t, S_t and R_t and represent the marginal capital values of these stocks. Maximizing the Hamiltonian and solving the resulting set of first-order conditions along with the constraints of the maximization problem gives an investment demand equation of the form

$$V_t = V\left[Z_t, S_t, R_t, P^t, W^t, \tau^t, U^t, \lambda_{T+1}, \mu_{T+1}, \rho_{T+1}\right],$$

where $P^t = (P_t, P_{t+1}, \ldots, P_T)$ and similar notation applies to other variables. Thus the optimal investment in each period is a function of the current stocks of capital and resources, current and future prices, and the terminal values of the capital and resource stocks.

Using the investment demand equation for V_t together with the equations of motion for R_t and S_t and the equation for output supply and input demand, one can solve for the long-run paths of all variables determined by the farmer. Note that the short-run and long-run models suggest a very different model of interaction between the economic, physical, and biological models. With the short-run economic model, economic decisions are made given the states of the physical and biological variables, and the physical and biological models are solved given the behavior of farmers. Time paths for the variables in each model are obtained by sequentially solving each model and using its results to condition the solution of the other model. In contrast, in the dynamic economic model, economic decisions are made taking into account the dynamics of the physical resource stocks and the population dynamics of species. Thus the time paths for the economic, physical, and biological variables are determined jointly in the solution of the dynamic economic model.

4. Mode Integration

4.1 Methodological Issues

Several methodological issues arise as the physical and economic model components are brought together into an integrated model. Successful integration requires compatible mathematical structures for numerical models and consistent statistical criteria need to be developed. In addition, several conceptual differences in model approaches exist across disciplines that need to be taken into consideration. The most important point to be emphasized in conducting this integration is the need for communication across disciplinary lines.

Physical versus Behavioral Modeling. First, there is a conceptual difference between the physical modeling, which relies upon physical constants, and behavioral models based on the assumed optimizing behavior of people. The structure of a physical model is invariant to changes in government policy, for example, but a model of farmer behavior may need to take into consideration the way farmers form expectations about policy. Consequently, the structure of a behavioral model may change over time as policy and other parameters change. The change in the structure of the behavioral model may in turn alter the linkages between the physical and economic models.

Experimental versus Nonexperimental Data. The physical and biological sciences rely primarily on data generated by controlled experiments. Economic analysis is generally based on nonexperimental data. Econometrics is devoted to the modification of classical statistical analysis so that valid inferences can be drawn from nonexperimental data. The differences in statistical methods need to be reconciled in the design of data surveys and research methodologies.

Modeling Approaches. Various disciplines find particular mathematical structures to be appropriate for their problems. For models to be integrated across disciplines, all disciplinary model components must be consistent with the ultimate goal of linking the models for policy analysis.

Selecting the Unit of Analysis: The Aggregation Problem. A basic methodological problem arises in any attempt to integrate the physical, health, and economic model components into a coherent whole; each component relates to a particular unit of analysis, each of which is generally different from the unit of analysis on which cost-benefit analysis should be based. The solution to this problem is to provide a statistical representation of the integrated model that can be defined over a common unit of analysis, and then to statistically aggregate to the unit of measurement meaningful to cost-benefit analysis.

4.2 A Statistical Approach to Model Integration

A key factor that needs to be taken into account in the modeling methodology is the heterogeneity of the physical environment and the related heterogeneity of agricultural production practices and associated environmental and health effects of those practices (Antle and Just, 1990). For example, an analysis of environmental fate of a pesticide based on a set of partition coefficients may be reasonable for a well-defined physical unit--say, 100 square meters of surface area--over which a specific set of parameters and input data are valid. But such a unit is generally much smaller than the economic or geophysical unit of analysis relevant to the assessment of social costs of chemical use. The relevant unit of analysis for social cost assessment may be as small as a farm or as large as an entire regional watershed.

To address the heterogeneity problem, an aggregate unit of analysis can be defined as a function of the problem context; e.g., for water quality problems the unit of analysis may be the land contained in a particular watershed. The land in the aggregate unit of analysis can, in turn, be disaggregate into sufficiently small units (plots) over which a valid set of physical and economic data and parameters can be defined. Associated with each plot is a vector of physical characteristics represented by $\boldsymbol{\omega}$. $\boldsymbol{\omega}$ may include physical characteristics such as depth to groundwater on the plot, the partition coefficients for the plot, the slope and elevation of the plot, and so forth. A stylized physical model can then be written C(X,w), where C is a vector of contaminant levels associated with the environmental partitions in the model (e.g., soil, air, water) and X is a vector of chemical applications.

As shown in section 3, a farmer's chemical-use decisions are functions $X(P, \psi, \tau, \omega)$, where P represents prices of outputs and inputs, ψ represents policy parameters, τ is technology parameters, and w is as defined above. Let the environmental characteristics of each plot of land in the region be fixed at a point of time and distributed across plots according to a distribution defined by a parameter θ. This distribution of environmental attributes induces a joint distribution for input use X, crop production Q, and contamination levels. Define this joint distribution as $\phi(Q,X,C \mid P,\psi,\tau,\theta)$.

4.3 Statistical Aggregation

The joint distribution ϕ provides a basis for statistical aggregation across the plots into quantities that can be used to conduct policy analysis at the aggregate level. For example, by integrating X and Q out of ϕ, a marginal distribution of contamination can be defined: $\phi(C \mid P,\psi,\tau,\theta)$. Using this distribution, the tradeoffs between, say, mean chemical use and groundwater contamination can be estimated. This information can be combined with valuation data to estimate the value associated with groundwater contamination. In addition, an aggregate pollution function can be obtained by taking the expectation of C with respect to this marginal distribution, and that relationship can be used for analysis of pollution policy (see Antle and Just, 1990).

To illustrate the statistical aggregation procedures, let X and w follow a lognormal distribution such that

$$\begin{bmatrix} \ln X \\ \ln \omega \end{bmatrix} \sim N\left[\mu, \Sigma \mid P, \psi, \theta\right],$$

where μ is a (2 x 1) vector of means and Σ is a (2 x 2) covariance matrix. It follows that C is a random variable and its mean and variance are functions of μ and Σ, which are in turn functions of P, ψ, and θ. Thus, for example, the population mean contamination level may be expressed as a function of the population mean level of chemical use. This relationship can be employed in policy analysis. For example, if a dollar value could be attached to a specified reduction in environmental contamination, these data can be used in cost-benefit analyses of policies to reduce pesticide use.

4.4 A Simple Economic-Physical Groundwater Contamination Model for Policy Analysis

This section describes an integrated economic-physical groundwater contamination model for policy analysis. The model is defined for a given chemical at a given location, such as a plot or field, which is homogeneous with respect to both physical and economic characteristics. It is based upon the models presented in sections 2 and 3.

A Physical Model

Following earlier notation, let

X = quantity of chemical

C = concentration of chemical x in groundwater

z = depth to groundwater

m = time for transport from surface to groundwater

r = fraction of chemical remaining after transport to groundwater

t = time period t = 0,1,2,...

h = half-life of chemical in groundwater

h* = 0.693/h

following the model presented in section 2.2, assume: the chemical does not move laterally in the soil or groundwater; it degrades according to first-order irreversible reactions; and the groundwater is uncontaminated at time t = O. Then:

(4) $$C_t = \sum_{k=1}^{t} x_k R_{kt}$$

where

$$R_{kt} = r \exp\left\{h^* \left[t - (m + k)\right]\right\} \text{ if } t - (m + k) > 0$$
$$= 0 \qquad\qquad\qquad \text{if } t - (m + k) < 0$$

Note that R_{kt} is interpreted as the fraction remaining at time t > k from application at k, including the effects of transport to groundwater and decay in the groundwater. The equation (4) is quite general and compatible with any specification of the coefficients R_{kt}. For example, R_{kt} could be specified more generally to embody the effects of lateral movement of groundwater.

An "economic" interpretation of equation (4) is possible. Since $R_{k,(t+s)} = R_{kt} \exp(h^*s)$, and $R_{k,t+s} = 0$ fors < m, C_t can also be expressed as

$$C_t = \exp\left\{h^*(m+1)\right\} C_{t-1} + x_{t-m} R_{t-m,t}\ .$$

Thus C_t can be expressed in the form of an equation of motion of a capital stock, $K_t = (1 - \delta)K_{t-1} + I_t$, where K_t is the stock, δ is the depreciation rate of the stock, and I_t is gross investment. Under this interpretation, exp {h*(m + 1) } represents the depreciation of the "stock" of contamination due to the decay of the chemical that is already in the groundwater, and $X_{t-m}R_{t-m,t}$ represents the gross investment, which in this model is the additional chemical that was applied at time t - m and leaches to the groundwater at time t.

An Economic Model

To illustrate the basic economic relationships, assume the simplest possible conditions: production of a single crop Q with a single variable input, the chemical X, on the given unit of land. The farmer chooses X to maximize profit π subject to the production process

$$Q = X^{\alpha_1} .$$

Solving the profit maximization problem

$$\max_{x} \pi = p\,Q - w\,x$$

gives

$$X = \left[\frac{1}{\alpha_1} \frac{w}{p} \right]^{1/(\alpha_1 - 1)} \tag{5}$$

Impact of Policy Changes on Groundwater Quality

Consider now a policy that sets $p_t = p^*$ for all $t > t^*$. We have the following relationships:

$$\left.\begin{aligned} \partial C_t/\partial X_{t'} &= 0 \text{ for } t - t' < m \\ &= R_{t't} \text{ for } t - t' > m \end{aligned}\right\} \text{ and } t' > t^* .$$

Hence the elasticity of C_t with respect to $X_{t'}$ is

$$\epsilon_{tt'} = X_{t'}R_{t't} / C_t \tag{6}$$

The elasticity of X_t with respect to p_t is, according to the model in equation (5)

$$\eta_t = 1/(\alpha_1 - 1), \text{ for all } t. \tag{7}$$

It follows that the effect of raising p permanently at time t* by the amount $\Delta p^* = p^* - p_o$ is

$$\Delta C_t / \Delta p^* = \sum_{k=1}^{t} \left(\Delta X_k^* / \Delta p^*\right) R_{kt}$$

which in point elasticity form is, in general,

(8) $$\xi_t = \sum_{k=t^*}^{t} \epsilon_{tk} \, \eta_k$$

and using (6) and (7) becomes

(9) $$\xi_t = \sum_{k=t^*}^{t} X_k R_{kt} / C_t (\alpha_1 - 1) \, .$$

These relationships are illustrated in the Figure 3 under the assumption that before t^*, $p = p_o$, and input use occurs at fixed time intervals. Under the baseline scenario, input use generates a relatively slow increase in groundwater contamination levels; when policy raises 'the price of the crop, chemical use levels increase and the rate of growth in contamination increases. Observe that before t*, contamination levels increased by the amount ΔC_t each period, whereas after t* + m contamination levels increase by $\Delta C_t^* > \Delta C_t$ each period (note the delay of m between the time the policy change is implemented and it begins to have an effect on groundwater quality because of the transport time). The elasticity ξ_t measures the percentage increase in C_t for each time period. Note that ξ_t is zero fort* < t < t* + m and is an increasing value thereafter.

The analysis of a policy which reduced p once and for all would be similar and would show that a reduction in input use levels would reduce contamination levels over time. Note, however, that the effect of the policy on groundwater quality would occur with a delay of m.

This simple example illustrates several interesting points. First, equation (8) shows that, in general, the effect of policy on groundwater quality is a function of all of the physical and economic parameters required to obtain ϵ_{tk} and η_k, whether these values are estimated from simple or complex models.

Second, suppose that chemical input use was sufficiently low such that C = 0 for all t < t* because all of the chemical degrades in the soil during transport (r = 0). Then a policy that induced an increase in chemical use would not affect contamination until input use reached the critical level at which r becomes marginally positive. Hence it follows that a policy that increases input use does not necessarily decrease groundwater quality.

More generally, input use will not be at constant intervals and market prices will be changing over time in response to policy and market conditions, and the time path of contamination levels will be much more complicated.

Finally, note that this model applies to a specific site. As discussed in the previous section, it can be assumed that the physical and economic parameters follow well-defined distributions in the watershed. This distribution, in turn, defines a joint distribution in the watershed for C, Q, and X. This joint distribution can be used to represent the watershed statistically as a unit and to conduct policy analysis. For example, it would allow statements to be made about the effect of a policy change on the expected (average) contamination level, or about the probability that contamination at any site in the watershed is less than or equal to a critical value, such as a maximum contamination level set by a risk analysis.

5. Conclusion

Benefit-cost analysis provides the foundation for developing a framework for integrating the various strands of disciplinary research needed to assess the environmental impacts of agricultural chemical use. The ability to predict the likelihood that a chemical applied at a specific point will end up in the groundwater enhances the economist's ability to devise location-specific policies for efficiently meeting pollution standards. In essence, by utilizing appropriate economic and physical models, it may be possible to overcome some of the "nonpoint" characteristics of the problem.

The data needed to identify accurately the potential for environmental impacts of chemical use are location-specific and chemical-specific. These information needs include the characteristics of the chemical and the physical environment that provide a basis for estimation of the chemical's mobility and degradation in the environment, and farm-level and field-specific production data that allow the farmer's chemical-use decisions to be modeled.

The heterogeneity of the physical environment means that chemical transport must be modeled at a highly disaggregate level. Thus, farmers' chemical-use decisions must also be modeled at a disaggregate level. Policy issues must be addressed at a more aggregate level, however. The bridge between these two levels of analysis is a statistical representation of the physical environment and the producer population which provides the basis for statistical aggregation from the highly disaggregate level required for physical models to the more aggregate level of policy analysis. The integration of physical and economic models reveals that, in general, the effect of technological or policy changes on environmental quality will depend on key physical and economic parameters. Considering the demanding data requirements of the integrated physical and economic analysis, a critical issue facing researchers is to identify minimal information sets needed to accurately estimate physical and economic parameters.

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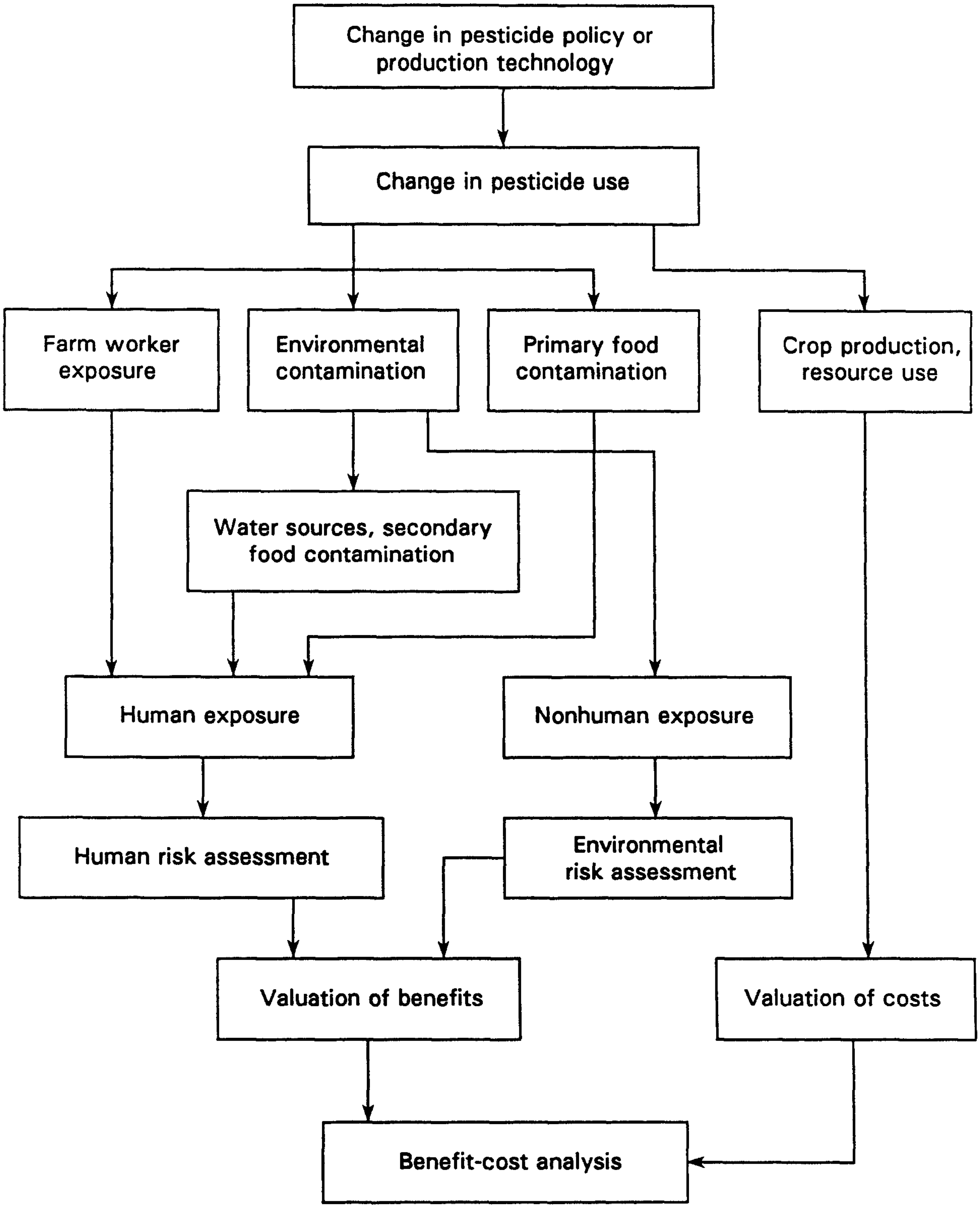

Figure 1. Major components of a benefit-cost analysis of a change in pesticide use.

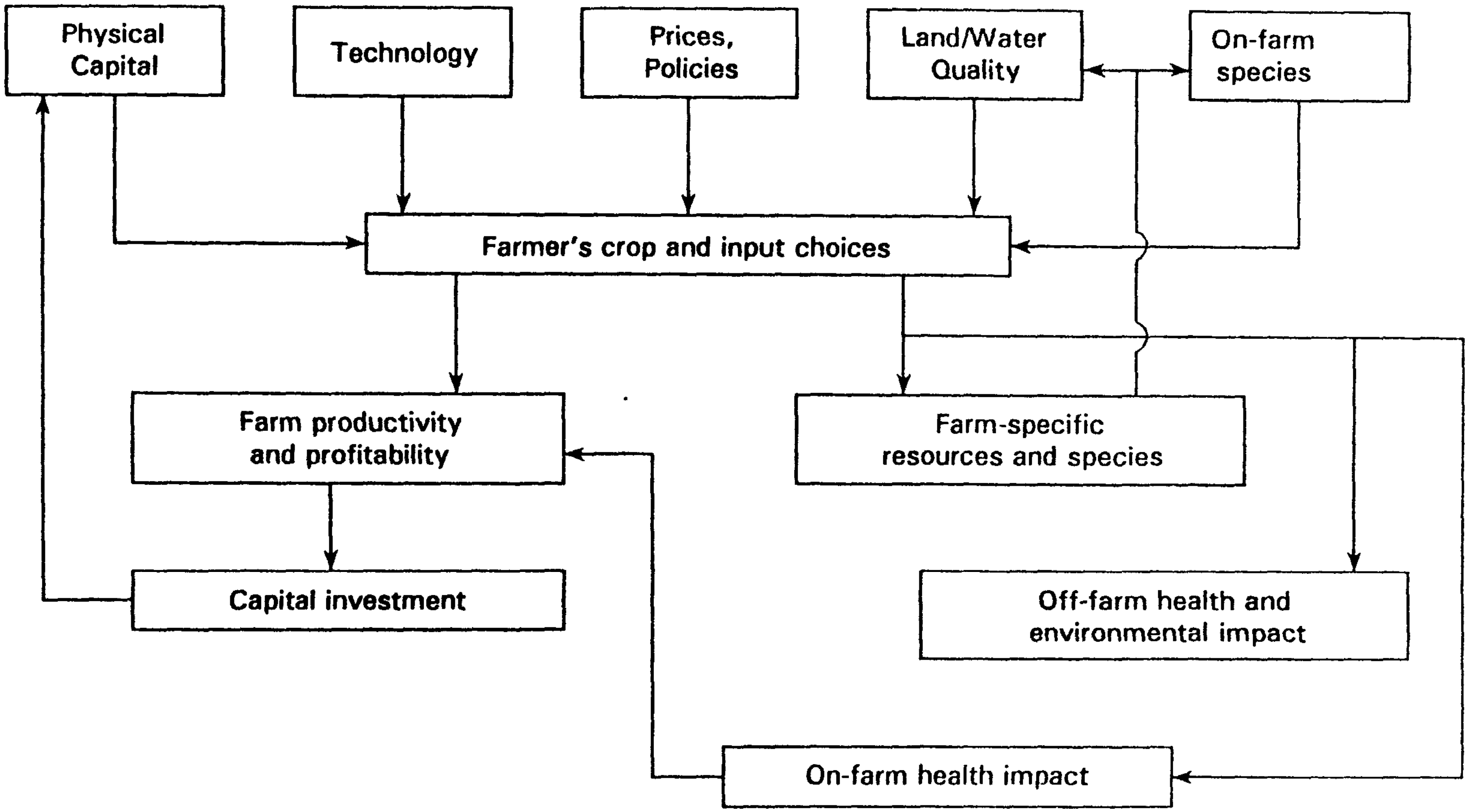

Figure 2. Production model linkages to on-farm resources and off-farm environmental and health impacts.

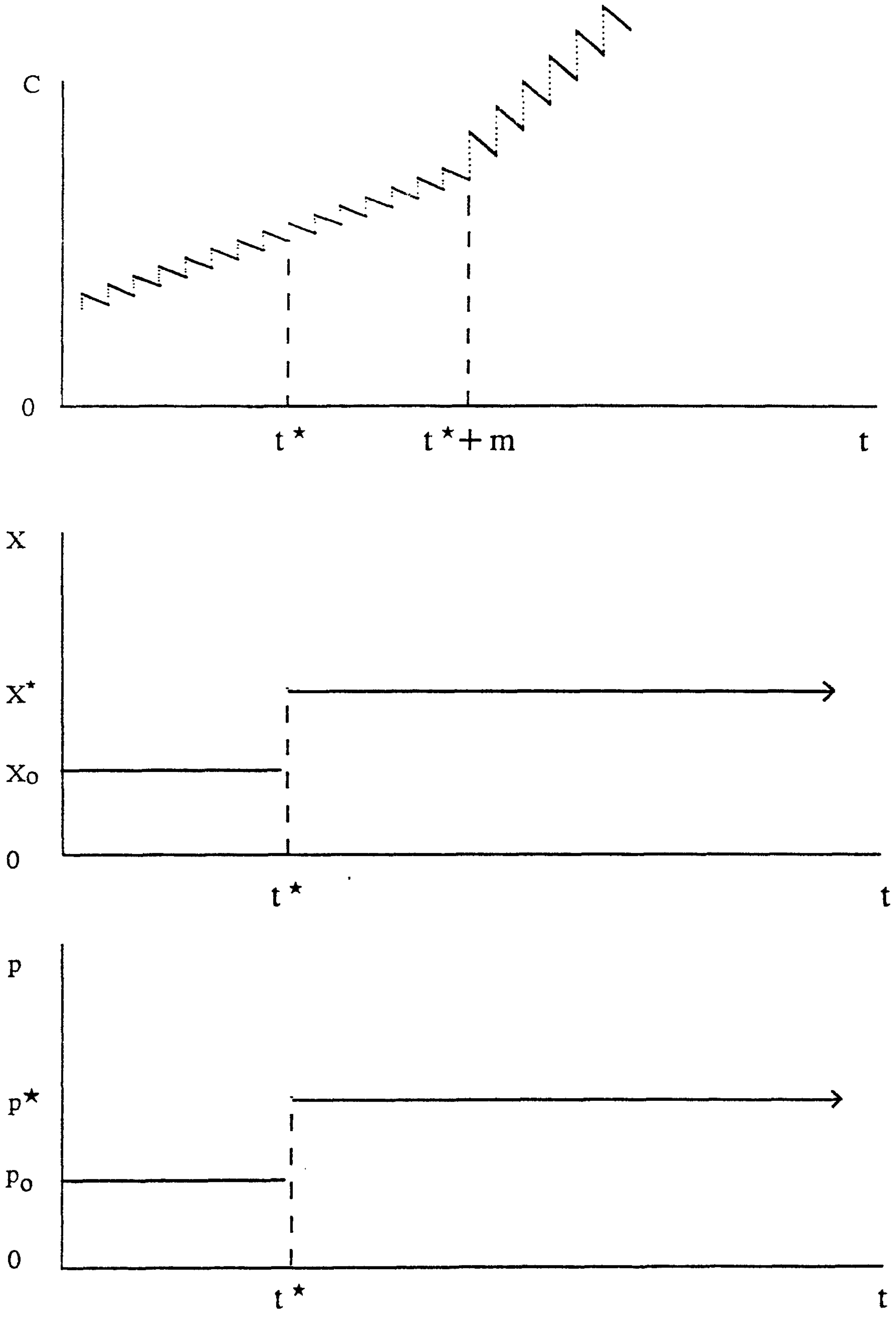

Figure 3. Time paths of output price (p), input use (X), and groundwater contamination (C) with a once-and-for-all change in price policy.

9

DATA REQUIREMENTS FOR MODELING AND EVALUATING NATIONAL POLICIES AIMED AT CONTROLLING AGRICULTURAL SOURCES OF NONPOINT WATER POLLUTION

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Since the inception of Federal water quality legislation (P.L.92-500, 1972), the aggregate control of nonpoint source pollution has been ineffective (GAO). Now several new national policy initiatives are in design or early implementation stages. States. are beginning to implement Section 319 programs under the 1987 Clean Water Act. The 1990 Coastal Zone Management Act mandates programs to reduce nonpoint pollution to coastal waters and authorizes regulatory approaches. Federal agencies are implementing the President's Water Quality Initiative to control largely nonpoint sources of agricultural chemicals leaching to groundwaters. And, just looming on the horizon is reauthorization of the Clean Water Act in 1992.

Despite incomplete theory and data, policy makers responsible for design and implementation of these national programs need analyses that cover the range of pollution conditions and potential economic effects. The challenge to economic researchers is to provide meaningful insights to the national policy process in the face of considerable scientific uncertainty. An aggregate evaluation of national policy alternatives should ideally possess several key features not always common to micro studies: endogenous prices, endogenous Federal program effects (e.g., agricultural commodity program participation), endogenous technology responses from the private and public sectors, regional tradeoffs in policy design, and complete government cost accounting. Ideally, these effects should be derived from a proper statistical aggregation as outlined by Antle and Capalbo, and by Opaluch and Segerson. But a comprehensive national data base necessary to perform such a sampling does not exist, and is unlikely to be built in the near future given budget constraints.

This paper examines the information requirements for modeling and evaluation of national nonpoint source pollution policies given scientific and data constraints. The planned economic evaluation of policies under the President's Water Quality Initiative1/ is used to illustrate the necessary analytical process. First, the basic policy-relevant questions guiding the data and modeling analyses are explored in some detail. Then, a preliminary modeling approach and data collection effort to address the static, short-run economic questions are described, including general model formulation. Possible empirical approaches and associated problems are presented. Future research priorities to enhance the policy relevance of economic analyses, such as induced technological change, are outlined at the close.

Focus of National Analysis

Three basic questions can be used to guide the economic investigations of national nonpoint source policies:

- What are the static and dynamic input and output changes from the policy initiatives?
- How do the input and output shifts map onto the natural resource base to produce positive or negative environmental effects?

1/ The Initiative is comprised of Federal programs of voluntary education, technical assistance and limited subsidies to achieve management practice changes that reduce potential agricultural chemical loadings, plus research and development programs to develop new technologies. Anticipated Federal expenditures are in the $400 - $500 million range over 1991-95. An evaluation of Initiative programs in comparison to alternative policies, such as regulation, is being directed the Economic Research Service.

- What are the economic costs to the private and public sectors, including program administrative expenses?

Input and Output Adjustments

Programs to control agricultural sources of nonpoint water pollution are designed to induce static shifts of inputs and outputs over space, and dynamic changes in production technology with positive environmental consequences. The desired end product is a series of static and dynamic input and output substitutions to reduce pollutant loadings to water resources. Ex ante modeling of the likely changes under alternative national policy approaches requires a clear delineation of many possible effects.

Perhaps the simplest starting point is short-run production behavior under profit maximization conditions (i.e., assuming a fixed total land base and technology). Potential substitutions of interest can be illustrated with a simple multiple input and output production relation (in notation consistent with Antle and Capalbo).

(1) $$T(Q, X, Z, R, S, \tau) = 0$$

Q is a vector of maximum rates of outputs with variable inputs X measuring labor, management, fertilizer, pesticides, etc., Z is a vector of fixed capital inputs including the total land base, structures, etc. , and parameter τ represents the state of technology. The production influences of physical resources, such as land and water qualities, are captured by R and biological organisms by S. The effects of input and output choices on R and S affect future production conditions and provide a dynamic production-environmental linkage. The production relation encompasses both intensive and extensive margin changes. How the inputs and outputs are jointly distributed over the environmental base then determines the nonpoint water quality consequences through time as Antle and Capalbo show.

Nonpoint water policies, such as the President's Water Quality Initiative, are mostly action program efforts including subsidized education and technical assistance (i.e, information) plus financial subsidies to shift the combinations of Q and X over space and time. Examples of these short-run (i.e., constant technology base) changes include reductions of leachable herbicides in favor of more management or mechanical tillage, and shifts in crop rotations to reduce nitrogen applications under highly leachable conditions. Estimating input and output substitutions thus becomes an important analytical focus.

The definition of appropriate input classes for estimating elasticities of substitution is a troublesome issue. For aggregate analysis, the input sets must be parsimonious. From an economic behavioral perspective, the classes should be substitutes in the decision-maker's mind. But to link the input changes meaningfully to the environmental relations, an economic class (e.g., corn herbicides) may need to be differentiated by leachability or half-life considerations. Obviously, a tractable aggregate analysis can not capture the full range of substitutions on all crops but should reflect the essential economic choices with basic environmental differences.

An important consideration to short-run output substitutions in agriculture is the role of Federal commodity programs. Through a system of program crop bases and differential deficiency (subsidy) rates plus acreage diversion requirements, the commodity programs bias the selection of crops relative to market conditions that might exist without commodity programs. Analyses of water quality policy effects on output choices must incorporate the roles of commodity programs and the potential competitive or complementary effects with nonpoint source control programs.

The discussion to this point has focused on the short-run economic effects, but nonpoint policies will occur in a dynamic, long–term context. Therefore, the values of Z and τ will vary from their fixed short run levels. And, the physical and biological variables, R and S, will change their temporal paths. In essence, the fixed factors, such as machinery types, and the production technology will likely change in response to private and public investment changes induced by alternative water quality policies. Antle and Capalbo explain the conceptual differences in time paths for the economic, physical and biological variables under short-run and longer-term, dynamic optimizations. For example, the President’s Initiative will invest in excess of one hundred million dollars to develop new technologies by public research agencies. Special attention needs to be paid to changes in relative factor prices caused by regulation, subsidies, or taxes that induce technological innovation. Both private and public sector research and development will likely be affected. Because little information exists to characterize these longer-term economic processes, that analysis poses data and modeling challenges as explored at the end.

Environmental Effects

Estimating how the input and output changes occur over natural resource conditions is necessary to predict the potential water quality effects. This estimation process is tractable at the firm or even watershed level, but becomes very complex when considering regional or national aggregate responses.

Opaluch and Segerson outline a conceptual procedure to join microparameter models (Antle and Just; Just and Antle) with geographic information systems (GIS) to characterize the potential water quality effects induced by an aggregate policy action. In brief, the process involves three basic steps:

1. Determining the water quality pollution potential of a microunit (e.g., field or farm)
2. Applying the microparameter model to characterize the extensive and intensive margin changes on the microunit due to the policy.
3. Determine the spatial distribution of environmental responses to reflect aggregate impacts on water resource units of interest (e.g., regional aquifers).

The authors note three potential problems with application of the linked microparameter - GIS modeling system. First, the microunit of analysis for the microparameter model and GIS must be reconciled. In most cases the appropriate decision unit for the microparameter model is smaller than available GIS data. Second, the microparameter models predict the response of a representative farm with certain characteristics but not the particular farm in a GIS cell. This problem can be lessened by aggregating the microparameter model results to a level (e.g., county) consistent with the GIS cell. Finally, available GIS data technology may necessitate a larger microunit (e.g., collection of farms) for analysis, but at the cost of sacrificing natural resource diversity affecting the specific nature of nonpoint water quality conditions. Despite the potential problems, the general approach appears to be the only feasible method at present of aggregating environmental responses for regional and national analyses.

Even a successful implementation of the linked microparameter - GIS model approach leaves two possibly important deficiencies in the environmental effects assessment. The methodology described by Opaluch and Segerson is largely short-run, static for both economic and environmental effects. Where longer-term, dynamic processes are important to nonpoint water quality policy responses, the microparameter and pollution potential algorithms should be altered to capture those effects. Second, the spatial and possibly temporal environmental responses are expressed in physical units rather than a common money metric. Thus aggregation of potential environmental benefits to regions of the nation are not possible due to incomplete science and data on fate-transport relationships and willingness to pay information.

Economic Costs

National policy makers are keenly interested in the economic costs of alternative water quality policies, both private producer and consumer welfare changes, and net public government expense impacts. Indeed, the government cost component has received increasing weight of late due to the large and continuing budget deficit. So credible estimates of the short-run and longer-term paths of economic and government cost components are critical to a national policy evaluation.

A bottom-up statistical aggregation to a *national* level of microunit cost supply responses using the microparameter model is impossible given current databases. Therefore the aggregate analysis of economic costs must necessarily proceed with large national models without explicit natural resource linkages. Such an approach introduces the possibility of inconsistent microparameter and aggregate estimates due to different model formulations. One approach to reduce inconsistencies is to use results from the micro level analyses to condition the aggregate modeling procedure. An example is to use the range of estimated elasticities of input substitution from the micro analyses to bound the regional responses induced by agricultural nonpoint water quality policies.

A short-run economic cost analysis requires the incorporation of several important factors. First, the effects of cost and supply changes on crop and livestock prices must be estimated including international trade impacts (i.e., output price endogeneity). The second round price repercussions of a national policy may complement or offset first round effects on microunits. Second, the analyses must permit static input and output substitution between all relevant factors of production and commodities to capture intensive and extensive margin changes under existing technologies. Third, the influences of existing and anticipated Federal agricultural commodity and conservation programs on inputs and outputs should be incorporated. For example, the effects of land diversions under the commodity program acreage set asides and with the Conservation Reserve Program will likely increase land prices and cause farmers to substitute non-land inputs such as chemicals (Offutt and Shoemaker). Finally, the cost analysis should capture the expected changes in government expenses, including water quality policy administrative costs and commodity program savings from reduced supplies and increased market prices.

The more challenging task is to extend the economic cost analysis to the longer-term. Two factors are critical to developing estimates of long-run economic adjustments. The changes in the fixed capital base to accommodate water quality programs are relevant. An example is a switch to more efficient irrigation equipment to increase use efficiency and reduce excess runoff and percolation. Induced technology diffusion and change as a result of water quality policies and/or changes in relative factor prices may be the most critical long run component. Ex ante economic analyses of policy impacts often greatly exaggerate the ultimate industry and economy wide impacts due to the static capital and technology assumptions. Longer-term elasticities of substitution for inputs affecting water quality are necessary to estimate the ultimate economic cost path.

Aggregate Modeling Framework

A full articulation of the relevant questions is a necessary first step in the national analysis. Unfortunately our ability to ask policy relevant questions is not matched by our capacity to capture those effects with available data and empirical methods. Nonetheless, a specification of a general aggregate modeling system is necessary to gain insight about how the feasible analytical approach differs from the ideal conceptual methodology. The modeling framework to follow focuses primarily on the economic input and output adjustments conditioned by resource characteristics and leaves aggregate environmental effects to research challenges discussed at the conclusion. The role of special data collection efforts, termed “Area Studies”, to enable the aggregate modeling analyses is then discussed.

To summarize, the key challenge of our research is to examine the relationships between the natural resource base and production activities for national policies. That is, how do different resource characteristics affect production decisions, and given those resource characteristics, how do production choices affect environmental attributes associated with those resources?

To formalize these questions, we present a general model to provide a conceptual basis for analysis. In what follows we describe a static general producer optimization problem and the associated loadings of pollution conditioned on regionally specific resource characteristics. A microeconomic model is developed retaining the essential microparameter concepts where individual producers face parametric prices and endogenous commodity program participation. Firms are then aggregated based on regional distributions of resource characteristics to market level commodity supplies and factor demands. Factor supplies are assumed to be perfectly elastic but commodity supplies face market level demand curves thus endogenizing commodity prices.

We allow the firm to be characterized as a multiproduct firm employing several inputs and producing several outputs to keep the analysis as general as possible. To restate equation (1), assume production by the j^{th} firm is determined by a transformation function represented as,

(2) $$T^j(Q, X, Z, R, \tau) = 0$$

where Q is a vector of outputs, X is a vector of variable inputs, Z is a vector of fixed factors, R is a vector of resource characteristics that contribute to production and pollution and τ is an index representing a particular technology.2/ It is the elements of R and τ that define input, output and resource linkages that are critical for the analysis. For example, if the firm is located in a dry climate on a sandy soil, it is possible that the firm will use a technology involving irrigation. The potential environmental damages derived under these conditions is entirely different from what might occur in a more moist temperate climate on a clay soil.

The above arguments make clear that a pollution loading function is also a function of the same arguments. That is, potential loadings will be a function of the outputs produced, inputs used, and R and τ. The pollution loadings function is expressed as,

(3) $$h^j = h^j(Q, X, Z, R, \tau)$$

Firms are assumed to be profit maximizers, and assuming the transformation

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2/ For ease of exposition, the physical and biological resource characteristics are collapsed into one vector, R, for the general analysis. The resource vectors should be divided into classes to capture the essential economic and environmental dimensions of the problem under study. Opaluch and Segerson suggest a three way classification resource characteristics, i.e., those affecting production only, production and pollution, and pollution only.

function obeys the usual properties, a profit function (abstracting from government programs) can be defined as,

$$(4) \qquad \pi(p, w, Z, R, \tau) = \underset{Q, X}{Max}\{p \cdot Q - w \cdot X : T(Q, X, Z, R, \tau)\}$$

where p and w are the output and input prices. Maximal profits and the envelope conditions yield optimal input demands and output supplies as the respective gradient vectors,

$$(5) \qquad X_j^* = \nabla_w \pi(p, w, Z, R, \tau)$$

$$(6) \qquad Q_j^* = \nabla_p \pi(p, w, Z, R, \tau)$$

The pollution loading associated with the optimal inputs, X_j' and supplies, Q_j' for the j^{th} firm is,

$$(7) \qquad h^j = h^j(Q^*(p, w, Z, R, \tau), X^*(p, w, Z, R, \tau), Z, R, \tau)$$

Loadings are indexed to the j^{th} firm to emphasize the point that loadings are specific to firm activity levels and the firm's resource characteristics. 3/

Commodity Program Participation

Output decisions and factor demands are affected by participation in commodity programs. The production incentives derived from support prices and requirements for program participation affect relative factor demands at the

3/ Indexing the H(.) functions by j is not meant to imply the functions differ over firms, rather it merely implies that there are multiple firms. This assumption could be relaxed if we treated τ as a random variable and then integrated over τ.

intensive margin and commodity supplies at the extensive margin. Producers choose to participate in programs based on the relative benefits and costs of program participation conditioned on their costs of production. That is, a high cost producer will more likely enter the program than low cost producers. The relative costs of production among producers are in part determined by the distribution of resource characteristics. Therefore we define a subset of the vector R to include variables that contribute directly to the productivity of firms and their ability to earn net returns.4/ We define ω to be a variable that determines productivity which spans the range $[0,\bar{\omega}]$, where $\bar{\omega}$ is the upper value of ω. Given market prices and program parameters there is a critical value, denoted $\underline{\omega}$, associated with net returns where producers begin to participate. Therefore, for values between $\underline{\omega}$ and $\bar{\omega}$, net returns are sufficiently low that producers will participate (given program parameters).5/

To keep things simple, we present a stylized version of programs. Program parameters are limited to a target or support price, $\bar{p}$, the set-aside rate, θ and the program yield rate, $\bar{q}$. Program benefits are determined as the product of the difference between the support and market price, $(\bar{p}-p)$, times land net of the set-aside and the program yield rate, $(1-\theta)A\bar{q}$. Producers choose to participate given the maximum profit of,

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4/ Here we are making the assumption that we can distinguish resource characteristics associated with program participation from other characteristics. While this is done for analytical convenience, it remains an empirical issue whether this distinction can be made.

5/ Program participation behavior could be estimated using a dichotomous choice model. Models of this sort have treated variables such as ω as unobserved. Within the current context, the variable may actually be observed.

$$(8) \qquad \pi = \begin{cases} p \cdot Q - w \cdot X \ : \ T(Q, X, Z, R, \omega, \tau) & \text{out} \\ p \cdot Q + (\bar{p} - p) \cdot (1-\theta) \cdot A \cdot \bar{q} - w \cdot X \ : \ T(Q, X, (1-\theta)A, Z, R, \omega, \tau) & \text{in} \end{cases}$$

where land has been identified separately from the vector of fixed factors, Z and is denoted A. "Out" refers to producers out of the program and "in" refers to those that are in the program. The cost of participation is the opportunity cost of setting aside land. The resulting profit functions are,

$$(9) \qquad \pi = \begin{cases} \pi(p, w; Z, R, \omega, \tau) & \text{out} \\ \pi(p, w; \bar{p}, \theta, \bar{q}, Z, R, \omega, \tau) & \text{in} \end{cases}$$

<u>Aggregation</u>

Aggregate or total industry supply and factor demands are found by integrating over the distribution of resource characteristics, R and ω.

$$(10) \qquad Q = \int_0^{\omega}\!\!\int Q^*(p, w; Z, R, \omega, \tau)\, dR\, d\omega + \int_{\omega}^{\bar{\omega}}\!\!\int Q^*(p, w; \bar{p}, \theta, \bar{q}, Z, R, \omega, \tau)\, dR\, d\omega$$

$$(11) \qquad X = \int_0^{\omega}\!\!\int X^*(p, w; Z, R, \omega, \tau)\, dR\, d\omega + \int_{\omega}^{\bar{\omega}}\!\!\int X^*(p, w; \bar{p}, \theta, \bar{q}, Z, R, \omega, \tau)\, dR\, d\omega$$

Total pollution loadings are found similarly as,6/

(12) $$H = \iint h(Q^i(\cdot), X^i(\cdot), Z, R, \omega, \tau)\, dR\, d\omega$$

where i indexes participants and nonparticipant. By totally differentiating equation (12) we can determine the information requirements necessary for modeling changes in aggregate pollutant loadings. The change in total loadings is expressed as,

(13)

$$dH = \iint \Bigg\{ \frac{\partial h}{\partial Q^i}\left[\frac{\partial Q^i}{\partial p}dp + \frac{\partial Q^i}{\partial w}dw + \frac{\partial Q^i}{\partial \psi}d\psi + \frac{\partial Q^i}{\partial Z}dZ + \frac{\partial Q^i}{\partial R}dR + \frac{\partial Q^i}{\partial \omega}d\omega + \frac{\partial Q^i}{\partial \tau}d\tau\right]$$
$$+ \frac{\partial h}{\partial X^i}\left[\frac{\partial X^i}{\partial p}dp + \frac{\partial X^i}{\partial w}dw + \frac{\partial X^i}{\partial \psi}d\psi + \frac{\partial X^i}{\partial Z}dZ + \frac{\partial X^i}{\partial R}dR + \frac{\partial X^i}{\partial \omega}d\omega + \frac{\partial X^i}{\partial \tau}d\tau\right]$$
$$+ \frac{\partial h}{\partial Z}dZ + \frac{\partial h}{\partial R}dR + \frac{\partial h}{\partial \omega}d\omega + \frac{\partial h}{\partial \tau}d\tau \Bigg\} dR\, d\omega$$

where $\Psi = \{\bar{p}, \theta, \bar{q}\}$. Equation (13) suggests how elements such as resource characteristics and technology have direct and indirect influences on pollution loadings. If we express the above total differential in elasticity form, we can see that the parameters needed for evaluation are mostly standard

6/ Of course, total pollution loading is an artificial construct in that pollution is defined by resource supply and demand. However, the aggregate pollution concept is useful to illustrate some aggregate production and environmental relationships of interest. Perhaps a reasonable example of this aggregate concept is the total leachable nitrates into groundwater aquifers.

producer behavioral parameters, i.e., supply and demand elasticities. We also assume that in the short run the resource and technology characteristics do not change, i.e., $dZ = dR = d\omega = d\tau = 0$.

The elasticity form is expressed as,

$$(14) \qquad \hat{H} = \iint \{E_{hQ}[E_{Qp}\hat{p} + E_{Qw}\hat{w} + E_{Q\psi}\hat{\psi}] + E_{hX}[E_{Xp}\hat{p} + E_{Xw}\hat{w} + E_{X\psi}\hat{\psi}]\} dR\, d\omega$$

where E_{ij} is the elasticity of j with respect to i and the "^" denotes percent change. From equation (14) we see that in order to determine the change in pollution loadings, (in the certainty case) given assumed changes in input and output prices, there are three basic (hard-to-come-by) types of information required. The first type includes the elasticities of demand and supply with respect to input and output prices and policy parameters (participation rates) . The second information requirement is the distribution of resource characteristics over the production space. Finally, knowledge of the fate and transport properties of various chemical inputs and soil profiles is needed to understand the relationships between inputs, outputs and loadings. The first information requirement represents an activity that economists have expertise in, and the second represents an important data collection exercise discussed below. The third requires knowledge of hydrology and geology, an area in which economists do not have a comparative advantage. Furthermore, the science that is developing in this area is generally limited to very small units of analysis, (e.g., fields or subfields) units that are below a relevant scope for policy analysis and also pose the problem that aggregation is impossible given present databases. Yet economists must work with physical scientist to insure that the fate and transport data are integrable with the economic analyses. For the present analysis, we limit our attention to agricultural production embodied in equations (10) and (11).

Equations (10) and (11) can be used to develop an aggregate economic model. Some of the key variables of interest are commodity supplies and prices and factor demands. Additional economic indicators are net income or rents and government outlays. An algebraic schematic of the model is presented in the appendix. The simulation model utilizes the aggregation methodology of Johansen (1972) and Hochman and Zilberman (1978). This aggregate model is presented to highlight the data needs for this kind of empirical analysis and to set the stage for a discussion of some data activities underway at the Economic Research Service.

The model could work as follows. We can think of the aggregation of firms as a grid, e.g. a collection of Major Land Resource Areas (MLRAs) or Area Studies (to be discussed below), or some other geographically defined regions. The acres of farmland in each region represent a percent or share weight of the total farmland. Within each region there is a distribution of resource characteristics. The distribution of resource characteristics condition commodity supplies, program participation and factor demands. Factor supplies are assumed to be perfectly elastic, therefore factor prices are treated as exogenous. Commodity supplies are aggregated according to their weights in each region and then are aggregated across regions according to the regional weights of the total acres of farmland. The model is closed with an aggregate commodity demand function which endogenizes prices. Commodity program participation could be modeled with a dichotomous choice model based on relative returns again conditioned on resource characteristics in the set ω. Given program participation, an accounting identity can be defined that determines government outlays.

Data Needs

The above model description highlights the data needs for this kind of analysis. The data can be categorized into two broad classes: (1) production data, e.g., input and output prices and quantities, and (2) resource characteristics. While there is certainly nothing unique about the first requirement for economic analysis, it is the scale of analysis combined with the second requirement that makes the data issues more demanding than usual.

Area Studies Project

USDA's Area Study project is, in part, a data collection effort designed to provide micro-level information on the relationship between agricultural production activities and characteristics of the resource base, information that is required by the model presented above. It is a practical matter that resources are not available to collect data on the full scope of agricultural production and natural resource conditions necessary to represent all categories of water quality problems related to agriculture. USDA is approaching the problem by selecting a set of "evaluation sites" such that the most important agricultural production and water quality combinations are covered. Not all contributions can be included because of limited data collection resources. Emphasis is placed on major field crops, such as corn, soybeans, and wheat, which rely heavily on chemical applications and cover broad geographical areas.

Specific objectives of the data collection component of the project are to:

1) Provide chemical use and farming practice information for selected National Water Quality Assessment (NAWQA) study sites to aid in understanding the relationship between farming activities and ground water quality for a variety of agroecological settings.

2) Sample a wide range of farming practices and resource characteristics using a consistent approach to provide for cross-comparisons and a comprehensive analysis of the national impacts of alternative policies.

A total of twelve Area Study sites will be investigated. Four areas have been selected for study in 1991—the Central Nebraska Basin, the White River (Indiana), the Lower Susquehanna Basin (Pennsylvania), and the Mid-Columbia Basin (Washington). Four new sites will be selected for study in 1992, and another four in 1993. Each of these areas corresponds to a USGS study site in the National Water Quality Assessment Study. This coincidence of study sites insures that analysis of the fate-transport aspects will be studied.

At each site, a chemical-use and farming-practice questionnaire will be administered to approximately 1000 farm operators. The location of the operator will coincide with a National Resource Inventory (NRI) sampling point. (The Soil Conservation Service conducts a National Resources Inventory every five years. The Inventory will be done again in 1992.) The NRI is based on a stratified random sampling design in which soil, water, and related natural resource data are collected at nearly a million sample sites. Choosing the sample so that it coincides with a NRI point insures that important information on soil properties will be available, and also provides a statistical basis for aggregation within the region.

The questionnaire will solicit information specific to the field associated with the NRI point and also for the whole-farm operation. Sufficient field-level data will be collected to describe in detail the cropping system used at the NRI sampling point (crop type, tillage practice, rotation scheme, chemical use, non-chemical pest control, etc.). More general whole-farm questions will be asked on acres planted by crop, chemical use by crop, general tillage practices used on the farm, and the size and type of livestock operation. Economic questions related to the whole-farm operation will also be asked to support development of economic models (such as the value of land, labor, and capital available to the operator and participation in government programs).

Possible Empirical Applications

The aggregate conceptual model described above requires bottom up statistical aggregation of the microparameter models. But the area studies data collection effort will fall short of the necessary coverage to perform that statistical aggregation for the nation as a whole. Two empirical approaches are possible recognizing the incompleteness of coverage.

Area study data can be used to estimate producer behavioral response functions (e.g., restricted profit functions, input demand, output supply) conditioned on the resource base. These area-specific supply and input demand functions would then describe an area-wide farm. A special challenge will be to estimate input and output substitution relationships with minimal cross-sectional input and output price variation. Given knowledge of the area study samples regarding input, output and resource relationships, the results could be extrapolated through application to other NRI points nationwide that match closest with the output and resource conditions studied. Such a procedure falls short of a proper statistical aggregation as outlined by Opaluch and Segerson on two counts. First, the area study models assume that firm-level behavior in relation to resource conditions can be approximated with one (or possibly two) field observations. Second, the extrapolation of estimated area study results to other areas based on output-resource matchings ignores possible technology variations across regions (e.g., fertilizer and pesticide practices).

The second approach is to capture essential aggregate and area-level production and environmental details in separate but linked analyses. The procedure would begin with the use of an aggregate (national) model of agricultural production and input use divided into major regions (e.g., collections of states). While the aggregate model is consistent with the above microeconomic-based analytical model, it does not include the explicit influences of the natural resource base. Important features of the aggregate model include price endogeneity, commodity program participation, output substitution and input substitution (relevant to water quality analyses). One candidate for the analyses is the US Agricultural Resources Model (USARM) (Konyar and McCormick). The USARM model does not have explicit natural resource detail since it uses aggregate regional production and cost responses. The area studies could be used to specify important input and output substitution relationships to provide some consistency between the aggregate and area study levels. In the second stage, the aggregate price shocks induced by policy shifts are entered into the area–level models along with other policy parameters (e.g., chemical restrictions) to simulate the net effects on output and input use in relation to the natural resource base. This second approach allows the area studies be separate investigations, but uses scientific insight from the survey analyses as both inputs to the aggregate model and as a mechanism to simulate aggregate level policy shocks. Extrapolation of the area study simulations to other regions based on common NRI output-resource pairings could proceed as in the first approach to estimate aggregate pollutant loadings and environmental shifts.

Future Research Priorities

The data and modeling approaches outlined are essential first steps mostly focused on the short-run economies, but do not cover longer-term or environmental issues. Areas for further investigation include induced technological change, fixed inputs, environmental effects, and government program expenses.

Technical Change

Economists recognize the critical and often complex roles of technology in resource and environmental management. Analyzing the impacts of environmental policy with a fixed technology set is rarely sufficient. The induced innovations literature has documented the role of relative factor prices in generating technology development and adoption (Hayami and Ruttan). Incorporating effective factor prices for non-market environmental services through public programs of subsidies, taxes, and/or regulation will likely induce technology shifts changing the longer term economic and environmental effects. Moreover, reform of commodity programs will likely change the technology stream. Two activities are planned to help incorporate the technical change influences. Studies of other environmental management programs will be consulted to determine if generalizations about technology response can be made for application to nonpoint water quality issues. Second, a Delphi technology assessment exercise will be conducted by interviewing public and private experts regarding emerging technologies relevant to nonpoint source control. Estimates of technical (input and output) performance, economics and environmental parameters will be obtained. Information from either source can be used to adjust input and output substitution relationships in the aggregate and area study models.

Fixed Inputs

Another dynamic process is the change in the short-run capital stock over time due to water quality policies. Examples include changes in pesticide or fertilizer application machinery and irrigation equipment. Antle and Capalbo present a long-run dynamic investment model wherein the farm chooses the sequence of investments to maximize present value of profit over the planning horizon. Conceptually, shifts in the fixed capital inputs change the parameter Z in the transformation (eq.2) and pollutant loadings (eq.3) functions which affects input demands, supply functions, economic costs, etc. Estimating Z endogenously requires knowledge of the investment demand structural equation and how that equation shifts in response to water quality policies.

Environmental Effects

Describing the impacts of national nonpoint policies on environmental resources may be the greatest challenge. As discussed, the area studies will be conducted in concert with USGS scientists to enrich the fate-transport analyses. It is unlikely that definitive information on the water quality effects of reduced chemical use will be available within the next decade.

Environmental process models can be used to describe changes in pollutant loadings at various points in the soil profile due to input-output shifts by water quality policies. Use of the NRI sampling points for the area studies provides critical physical resource information for the process models, including soils data, precipitation, and other variables. When these data are joined to estimated input and output changes from the area study behavioral models, then geographical summarization of the pollutant loadings can proceed along the lines advanced by Opaluch and Segerson. The estimation process would describe comparative static outcomes but not the dynamic path of pollutant change.

Valuation of the environmental effects is equally problematic. Given uncertain fate-transport knowledge and virtually no epidemiological data, objective exposure and health effects modeling is not feasible. Two approaches will be explored. First, for those water systems estimated to exceed maximum acceptable contaminant levels by survey data or process model extrapolations, the cost of obtaining alternative water supplies can be calculated as a minimum bound. The second approach is to elicit willingness to pay estimates through contingent valuation exercises.

Government Program Costs

With few exceptions, most studies of environmental policies ignore the roles and magnitudes of public expenditures. Though the costs are often transfers, their influence on decision making is important. For the President's Water Quality Initiative based on large scale eduction and technical assistance programs, government expenditures will total hundreds of millions of dollars. With a continuing Federal deficit problem, the minimizaiton of those expenses is an important objective. Estimates of the program costs will be assembled based on experience in demonstration and special water quality projects conducted under the Initiative. Estimates for other water quality policies in comparison to Initiative programs will be made based on Federal or State environmental policy experience or engineering projections.

Concluding Note

The evaluation of national water quality policies poses some very special data and modeling problems. Survey funds are not available to do comprehensive data collection consistent with theoretically-based microparameter models for a bottom-up aggregation to a national level. However, it appears possible and desirable to incorporate some micro-level detail, especially on production-resource economic and environmental linkages, into the aggregate framework. Longer-term issues of incorporating technological change, capital stock changes, and portraying aggregate environmental effects are important research agenda items.

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Appendix

The following is a skeleton representation of an aggregate model of agricultural production spatially distributed over various resource characteristics and commodity program participants.

(A1) $$Q^S = \iint [\Omega Q^{in}(\cdot) + (1-\Omega) \cdot Q^{out}]\, dR\, d\omega$$

(A2) $$\Omega = g(p, w, \overline{p}, \theta, \overline{q}, \omega)$$

(A3) $$Q^D = D(p)$$

(A4) $$X^D = \iint [\Omega X^{in}(\cdot) + (1-\Omega) X^{out}]\, dR\, d\omega$$

(A5) $$Q^S = Q^D$$

(A6) $$G = \iint [\overline{q}(\overline{p}-p)(1-\theta) A^{in}(\omega) + WQT]\, d\omega$$

Equation (A1) is total commodity supply integrated over all resource types and commodity program participants and nonparticiapnts. Equation (A2) is a dichotomous choice function which determines the commodity program participation rate, Ω. Aggregate commodity demand is represented by equation (A3). Total factor demands are represented by equation (A4), again weighted by participants and nonparticipants. Equation (A5) defines market equilibrium. Total government outlays, including cost-sharing or other water–quality transfers (WQT) is calculated in equation (A6).

Regional Modeling and Economic Incentives to Control Drainage Pollution

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Regional Modeling and Economic Incentives to Control Drainage Pollution

Introduction

Management of quantity and quality of irrigation water--both as an on-farm production input and as an off-farm agricultural drainage residual--is an increasing concern in many parts of the world, including the arid western United States. Agricultural drainage water often carries salts, pesticides, nitrates, selenium, and other trace elements that pollute soils, surface water resources, and aquifers. As a non-point source of pollution, agricultural drainage water directly and indirectly affects agricultural productivity, wildlife, public health, and amenity resources. In addition to the quality aspects, strong competition exist for water among urban, industrial, environmental, and agricultural users in western United States. Water conservation in irrigated agriculture may achieve the dual goal of extending fresh water supplies and improving environmental quality.

Identifying solutions to the irrigation water quantity/quality problem involves two challenges. One challenge concerns the complexity of modeling the relevant physical and biological systems and their relationships to economic decisionmaking. These systems include both spatial and dynamic dimensions. Economic decisions involve private decisions, such as investment in irrigation technologies and land use on the farm, and collective decisions, such as the optimal sizing, siting, and timing of joint treatment facility for drainage water at the regional level. The second challenge concerns the design of an economic incentive system that will, simultaneously, provide socially efficient solutions and be acceptable to all interested parties.

The economic literature on irrigation water quantity/quality problems has expanded tremendously in recent years. It includes feasibility studies of technologies to reduce pollution as well as modeling and policy analysis.

Various management strategies to limit irrigation-induced water quality problems are being evaluated at the field, farm and regional levels in the San Joaquin Valley (SJV) of California. Improved agricultural management to reduce the quantity of drainage for disposal may involve cropping pattern adjustments, changes in water application rates for a given crop-technology, adoption of water conserving technologies, and management practices, and adjustment of irrigated acreage base (SJVDP, 1990).

The use of evaporation ponds, which under certain conditions may reduce environmental damages through reduced drainage disposed directly to the environment has been evaluated by Ford (1988). Dilution of drainage water with freshwater prior to disposal was analyzed by Stroh (1991), and reuse of agricultural drainage water has been examined by Rhoades and Dinar (1991). Biological and chemical treatment of drainage water for selenium has also been considered by Stroh (1991).

Currently, treatment procedures involving evaporation ponds, dilution, and chemical/biological treatment have been technically evaluated in many places. Findings suggest the existence of economies of scale in the construction and operation of treatment facilities (see also Klemetson and Grenney, 1975; Ergas et al., 1990; CH2M HILL, 1986; Hanna and Kipps, 1990; Gerhardt and Oswald, 1990).

An understanding of the effects of irrigated agriculture on soil and water resources is essential to an appropriate economic analysis of alternative. However, modeling the relationships between agricultural activity and the physical environment in which agriculture occurs is very complex.

To demonstrate the complexity of the modeling task, Figure 1 provides a scheme of applied water and drainage relationships. These relationships involve multi-space and time dimensions, and third party effects. Figure 2 presents the area of drainage related problems in the SJV. Is one physical model appropriate to address this issue for an area of over almost 3 million acres? Of course not. Wide variability in physical conditions suggests that agricultural effects on the environment may vary substantially. Figure 3 highlights this variability for two locations in the SJV: the position of the geological formations, soil type, depth and thickness of the corcoran clay, land slope, depth to a confined aquifer, distance to a river for drainage disposal, and many other factors prevent us from relying on one general physical model to adequately address the problem. Even if one could model these relationships accurately, there are still concerns availability of substantial resources required for data development and processing of simulations.

Policymakers depend on the research community to provide them with models that areas relevant as possible, as well as tractable and manageable. This often requires a simplification of underlying physical relationships. The following models while less exhaustive in scope and detail then some earlier efforts, may serve as useful tools for policy analysis addressing drainage and related problems in a specific region of the SJV.

Modeling efforts have included both site-specific models based on detailed empirical relationships estimated from local information and more general models that can be calibrated for local conditions. The former approach often involves development of robust multidisciplinary models that include all possible components (agricultural-hydrological-economic). One example is the attempt by the **SJVDP**[1] to develop the Westside Agricultural Drainage Economic (WADE) model. The model was supposed to simulate policy effects on physical variables and economic behavior of farm operators in 181 "cells" in the SJV, but was heavily dependent on local data which therefore limited its application (Hatchett et al., 1991; Imhoff, 1991). Another example involves a simulation model developed by Gates and Grismer (1989) that must be run on a super computer.

A second approach is to develop relatively simple models that address important aspects of the problem to analyze limited policy scenarios. The success and usefulness of such models for policy analysis is dependent on the ability of their developers to identify and model the essential problem components. An example is the model in Caswell et al. (1990) that consists of a limited number of state variables and contains simple relationships.

Given the complexities of modeling a physical/economic system, and the data and software limitations, a third approach might involve the use of several models, each emphasizing a different aspect of the problem for a given location. Then, models can be combined for policy design and analysis purposes.

This paper provides an example of this combined approach. The paper proceeds as follows. First, physical modeling will be discussed in relation to alternative types of models. Then impact models and policy design models will be compared. In both cases, models are applied for conditions in the SJV. Technical and empirical data of agricultural activity and water pollution in the SJV are based on previous work (SJVDP, 1990; Swain, 1990; and sources cited in Dinar et al., 1991a,b,c) and so will not be extensively discussed here.

Modeling physical-economic relationships

Physical-economic models may have two purposes: impact analysis and policy design. In an impact analysis, a policy maker may wish to determine the effects of some policy such as tax on water quality, or quota on water quantity, or requiring that certain new technologies be adopted. Because of political considerations, effects of interest would include not only water

[1]**SJVDP** or San Joaquin Valley Drainage Program (1985-1990) was formed to address drainage and related contamination in the Kesterson Reservoir and other locations on the westside of the San Joaquin Valley

quantity and quality determinations, but also economic impacts such as costs and profits for producers who would pay taxes and/or adopt new technologies. In a policy design mode, a policy maker would want to know what would be the best type of tax (e.g., on water quantity or on water quality) or what type of technologies might be recommended. Obviously, for this second type of analysis, impacts would also be important to know, but they would be a part of an optimization problem.

In both cases, physical models are important. In the policy design case, mathematical equations describing the physical system become the constraints in an optimization model. In the impact case, the economic decisions of producers in response to potential policies determine physical effects.

Below, an example of each of these types is given. in the case of impact analysis, models are of two types: steady state and dynamic.

The impact models are used to assess profit maximization responses of farm operators under conditions of water scarcity, low input quality and externalities; and to evaluate incentive programs, taxes, and quantity-based restrictions as alternative methods of achieving policy objectives; The policy designmodel is used to design regional cooperation in water resource use, drainage reduction, and treatment to reduce pollution.

The steady state regional model of agricultural water use and drainage water quantity/quality is developed by integrating physical, biological and agronomic models for the region (Letey and Dinar, 1986) within an economic decisionmaking framework (Dinar et al., 1990: Dinar et al. 1991b). Efficiency of technical solutions is evaluated relative to urban and environmental constraints on water quality and quantity. The farm-level dynamic model is developed that considers the effect of present decisions on future outcomes. The model evaluates the effectiveness of water use technologies for irrigation, water quality mixing, drainage treatment, and other farming practices to meet water quantity and quality constraints, including the demand for water by competing sectors. The policy design framework for inducing regional cooperation uses physical-economic models to consider incentives and cost sharing schemes required for adoption of appropriate technologies, both at an individual producer and regional levels.

A steady state modeling frameword

Consider a region with a given number of farms, each having a limited homogeneous area of productive land and a limited amount of irrigation water supply (surface and ground water) with known salt concentrations. The farmers are served by a water district (from hereafter district) which has a long-term federal contract to receive a certain amount of surface water annually, for a given price. The farmers pay this base price plus a "district charge" to cover delivery, maintenance, and overhead costs, and costs of drainage treatment provided by the district.

A number of alternative crops can be grown on each farm and these can be irrigated with different combinations of water quantity and quality. Subsurface drain tiles have already been installed in farms where shallow ground water and drainage problems affect farming, Therefore, the installation of tiles will not be considered a decision variable. The district collects drain water from sumps on each farm for treatment and disposal. The disposal outlet is constrained in both total volume allowed and quality (salinity). The district may use part of its surface water allocation to dilute drainage in order to meet the quality constraint.

Several on-farm and district-wide management options to reduce the agricultural drain water quantity and/or quality will be evaluated here. Most of these options have been considered and described at field and farm levels (Knapp et al., 1986), and for regional planning purposes (SJVDP, 1990). They include reducing irrigation rates, changing cropping patterns, improving water application uniformity (management and equipment), and treating drainage water. Individual farmers and the district can select one or more of these options in response to policy measures.

The model presented here is a steady-state one. It is assumed that the optimal solution is found relatively early along the planning horizon and that once it is found, it will be followed by the farmers and the district for the entire time horizon. Therefore, it optimizes decision variables for only one year, including all long-term economic costs related to the agricultural production process.

The regional model is designed to maximize regional net income:

$$[1]\ \mathrm{Max}\ R=\sum_i P^y_i \sum_j y_{ij}X_{ij}-\sum_j\sum_i M_{ij}X_{ij}-P^d\sum_j D_j\sum_j\sum_i K_{ij}$$
$$-\sum_j P^g_j\sum_i G_{ij}-\sum_j\sum_i P^s_{ij}S_{ij}-P^{ds}S^d-T_j\sum_j D_j$$

Here i is an index for crop and j is an index for farm. R is the regional net income, $P^y{}_i$ is the crop price net of harvest and marketing cost, M_{ij} are the per unit area variable production costs net of water related cost, P^d is the cost of pumping the drainage water, $P^g{}_j$ is the cost of pumping ground water, $P^s{}_{ij}$ is the price of water to the farmer, and P^{ds} is the cost of diluting the drainage water to be discharged (assuming no additional dilution cost except flesh water price to the district). Assuming that the difference $\sum_i\sum_j(P^s ij - P^{ds})\cdot S_{ij}$ **equals** the overhead, them these two components should not be included in the objective function. All revenues raised by increasing prices or taxes are assumed to be rebated to the farmers in a way unrelated to surface water used or drainage produced.

The above objective function is maximized subject to several constraints. These constraints are presented and explained in detail in Dinar et al.(1990). In the following only several model equations will be explained.

Relative yield (f_{ij}), deep percolation volume (d_{ij}), and salt concentration in the deep percolation water (z_{ij}), are functions of the quantity (a_{ij}) and quality (c_{ij}) of applied water, the application uniformity (u_{ij}) measured by Christiansen Uniformity Coefficient (CUC-used as a measure for the irrigation technology), and climatic conditions expressed by pan evaporation during the growing season (e_{ij}).

$$y_{ij} = f_{ij}(a_{ij}, c_{ij}, u_{ij} \mid e_{ij})\cdot Y_{ij}$$
$$[2]\quad d_{ij} = g_{ij}(a_{ij}, c_{ij}, u_{ij} \mid e_{ij})$$
$$z_{ij} = h_{ij}(a_{ij}, c_{ij}, u_{ij} \mid e_{ij}).$$

The variable y_{ij} is used here to express absolute crop yields. Y_{ij} represents the maximum potential yield that a given farm can achieve under optimal conditions, and reflects differences in management other than those considered in the production function. The pan evaporation variable allows the model to be transferred to any location (Letey and Dinar, 1986).

Irrigation water that infiltrates the soil is used in evapotranspiration or lost to deep percolation. In some areas, an almost impermeable layer of clay impedes the percolation of water (see also Fig. 1). This water collects and must be drained away to maintain productivity. Part of the drainage may occur as subsurface lateral flow to adjacent fields or farms that presents externality problems. The total amount of drainage water produced on farm j is

$$[3]\quad D_j = q_j \sum_i d_{ij} X_{ij} - \sum_{n \neq j} \beta_{jn} q_j \sum_{k \neq j} \mu_{jk} D_k$$

where q_j represents the severity of the drainage problem on farm j, ($0 \leq q_j \leq 1$); $q_j = 1$ means that all deep percolation results in drainage. It is assumed that each farm has homogeneous soil properties, so q_j represents on-farm drainage conditions. Parameter β_{jn} ($0 \leq \beta_{jn} \leq 1$) is the fraction of drainage produced on farm j that arrives at farm n ($n \neq j$; $\sum \beta_{jn} = 1$), and μ_{jk} ($0 \leq \mu_{jk} \leq 1$) is the fraction of drainage from farm k that arrives at farm j ($k \neq j$; $\sum \mu_{jk} = 1$). Subsurface lateral flow is one source of externality effect within the region. Where there are no lateral drainage flows, $\beta_{jn} = 0$ and $\mu_{jk} = 0$ for each n and k.

A quality (salinity) standard (C^d) may be imposed on discharged drainage. If the salinity exceeds that standard, the district must dilute the drainage with fresh water of a better quality. The quality constraint is:

$$[4]\quad \frac{\sum_j \sum_i z_{ij}D_j + S^d C^s}{\sum_j D_j + S^d} \le C^d$$

where S^d is the amount of surface water with a given quality C^s (C^s is a higher quality than z_{ij}) used by the district for dilution. Both the quantity and the quality constraints (as a matter of fact the product $D^d C^d$) reflect the assimilative capacity value that society assigns to the water body. However, quality standards and regulations on drainage pollution were commonly associated with one of these components only.

Each farm has an annual quota (S^f_j) of fresh water (also of quality C^s) provided by the district. Farms can supplement their surface supply by pumping ground water. The amount of ground water that each farm can use is constrained by pumping equipment. The district has no control on the annual amount pumped by each farm. It is assumed that ground water is pumped from a confined aquifer and does not affect the shallow water table.

The model currently assumes drainage salinity equals deep percolation salinity. In reality, the existing shallow ground water acts as a buffer, so that changes in deep percolation quality are only partly matched by changes in drainage quality.

The amount of irrigation water used on each farm is

$$[5]\quad \sum_i a_{ij}X_{ij} = S_{ij} + G_{ij} + R_i X_{ij} \qquad ,\forall j$$

where R_i is the seasonal effective rainfall for crop i, G_{ij} is ground water and S_{ij} is surface water applied.

The salt concentration in the irrigation water applied for crop i is

$$[6]\quad c_{ij} = [C^s S_{ij} + C^g{}_j G_{ij}] \,/\, [S_{ij} + G_{ij} + R_i X_{ij}] \quad \text{for each } j$$

where $C^g{}_j$ is the salt concentration of ground water in farm j.

One of several on-farm decisions is the type of irrigation technology used on crop i in farm j. In this model, uniformity of applied water is used as a surrogate for irrigation technology and irrigation management activities, with a more advanced technology being associated with a higher CUC value. Higher CUC values are associated with greater costs in irrigation hardware and/or management. The total irrigation cost (except for the cost of the water) is

$$[7]\quad K_{ij} = r_{ij}(u_{ij})\cdot X_{ij}$$

where K_{ij} is the annual irrigation cost for crop i on farm j. It is assumed that $\partial r/\partial u > 0$ and $\partial^2 r/\partial u^2 \ge 0$. That is, the cost of achieving a better irrigation uniformity application is increasing. Also it is assumed that $\partial K/\partial X = 0$; that is, no economies of scale are assumed with regard to the size of the irrigated field.

The district's annual quota of fresh surface water is allocated to farms and used to dilute drain water. The district purchases surface water for a given price per unit volume (P^{ds}) and then provides it to the farmers for a given price of P^s_{ij} per unit volume. The model allows the district to discriminate among farms and crops. The district may try to control water consumption by increasing and decreasing this price. In addition, the model allows the district to charge either a flat rate or a tiered rate for water.

$$[8]\quad P^s_{ij} = \begin{cases} P & \text{for } S_{ij}/X_{ij} \le H_{ij} \\ v(S_{ij}/X_{ij}) & \text{for } S_{ij}/X_{ij} > H_{ij} \end{cases}$$

where P^s_{ij} is the price per unit volume of surface irrigation water applied on crop i in farm j; H_{ij} is a parameter determining the maximum amount of water per unit area of crop i in farm j that will be charged the basic rate. ($P \geq P^{ds}$ and includes only the overhead of the district). The function v has a positive first derivative with regard to the per unit area water volume.

The district can also impose (or relay) a tax (T_j) on volume of drainage created by each farm. This is done assuming that the district monitors each farm's outlet and that the monitoring costs are either zero or are already included in the district services charged to the farms.

Additional technical constraints include available land, idle land, quantity of disposed drainage water, annual surface water allotment, and ground water pumping capacity. Idle land can also be adjusted to represent land conversion to non-irrigated uses.

The model was applied to a particular water district on the west side of the SJV. The water district is comprised of 12 farms; for simplicity, this analysis is concerned with three farms. Data on cropping patterns, prices, costs yields and water quality (Dinar et al., 1990). While surface water is the primary source of irrigation water, ground-water pumping is used occasionally by the farms to augment irrigation supply. For the purpose of the analysis it is assumed that unlimited ground water is available. Water quality inputs were set at 450 ppm (EC[2]=.7) for surface water and 1280 ppm TDS (EC=2.0) for ground water for all farms. Quadratic functions for yield and deep percolation volume and quality (salinity) were estimated by crop using the model suggested by Letey and Dinar (1986). District farmers use primarily

[2] 1 EC (mmhos/cm) = 640 ppm Total Dissolved Salts.

surface irrigation--furrow and border strip--with a current CUC of about 75. Improvement in irrigation technology are represented by increases in CUC. Irrigation technology cost functions were estimated for each crop using the cost data from CH2M HILL (1989) the irrigation technology CUC values from University of California Committee of Consultants (1988). Potential yield levels for each farm were estimated using the procedure suggested by Knapp, Dinar and Letey (1986), based on yield data obtained from the district for 1987-1989. Estimated coefficients for crop yield, drainage quantity, and drainage quality functions, exponential irrigation cost functions by crops, variable and fixed production costs (excluding water), crop yield prices and weather data for the different crops can be found in Dinar et al. (1990).

The model was used to assess alternative strategies to restrict environmental pollution while maintaining agricultural production. While several policy instruments were evaluated (Dinar et al., 1991b), only two will be presented. The first involves a tax on discharged drainage. Values used varied from $0 to $40/ha cm,[3] where $0 represents the “no regulation” case. The second policy instrument involves a flat increase in surface water price. Values varied from $0 to $3/ha cm, and are based on actual water price increases charged in a neighboring district under a similar policy (Wichelns, 1991). For simplicity, administration costs associated with the programs are not considered. Also at this stage, environmental costs are not included.

The “no regulation” case is represented by policy values of $0 for both water price increase and drainage tax. The base situation was simulated using the value of CUC=75, representing the current technology level.

The farms differ in their cultivated land area, fraction of applied irrigation water resulting in drainage, and also potential levels of different crop yields (Table 1). Farm 2 produces the highest drainage fraction and farm 3 the lowest drainage fraction for all levels of applied water. Therefore, it is expected that farm 2 will be more sensitive to drainage tax relative to the other farms.

Table 2 presents regional level results. Regional net income is defined as regional income plus the amount of collected taxes. Water and drainage taxes collected in the district must be redistributed or re-invested locally, as districts are not allowed to accrue profits (this is further addressed in the last section). In the case of a drainage tax, net income drops linearly (Figure 4) from $1.15 million to $.7 million with increases in drainage tax of $0 to $40 ha cm. In the case of an irrigation water tax, net income drops exponentially (Figure 5) from $1.15 million to

[3] 1 acre foot = 12.35 ha cm

$.7 million with increases in water price of $0 to $3.5 per ha cm. The share of taxes in the regional income varies from .54 to .60 at the higher tax levels as compared to .10 to .20 at the lower tax levels.

In general, farmers respond to increased water prices and drainage taxes by reducing surface water application rates, reducing cotton acreage, and increasing the rate of ground to surface water use. Farmers are less likely to invest in improved irrigation technologies (Table 3), but instead to reduce applied water by either cutting back on irrigation water rates or reducing cropped acreage. The reduction in the average water application per unit land is not significant in the case of a surface water tax (116, 115, 108 ha cm for 0, 3 and 6 $/ha cm) but very substantial in the case of drainage a tax (116, 79, 92 ha cm for 0, 10, and 40 $/ha cm). Similarly, acreage reduction is more significant in the case of a drainage tax (36%) than in the case of a water tax (22%).

The effectiveness of the two policy tools to reduce irrigation drainage pollution can be evaluated using the information in Figures 6 and 7. In the “no regulation” drainage volume discharged was nearly 65 thousands ha cm with a salinity concentration of nearly 8.5 EC. Drainage volume is reduced with both policy instruments. However, concentration of pollutants in the drainage water increases (Letey and Dinar, 1986), and pollution load to the environment (the product of pollution volume and pollution concentration) decreases and then increases as taxes increase. This is due to two effects: (1) as farmers reduce water application pollutant concentrations rise exponentially, and (2) in the case of tax on irrigation water, farmers replace surface water with ground water of lower quality. How society measures pollution is, therefore, essential in evaluating the success of policies.

A dynamic modeling framework

While a steady state model may provide useful insights on the impacts of irrigation-induced pollution, the dynamic nature of drainage and salinity pollution are not addressed. Salinity and other toxic accumulation in soil and water bodies has a direct impact on the quality of the resource base over time. Effects of present production decisions on future opportunities may be significant and therefore, should be included in a full analysis. Moreover, many resource policies are time dimensional (e.g., phased reductions in water supply) and are better handled within a dynamic framework. Unfortunately, dynamic relationships are often relatively complex and empirical estimates may be lacking (Knapp et al., 1990). Although an optimal steady state solution may be reached after 3-5 years under certain boundary conditions (Dinar and Knapp, 1986; Yaron et al., 1982), a dynamic framework is preferable.

A dynamic model of a farm-level (or regional homogeneous) operation is described in this section. The objective (Eq. 9) is to choose over time horizon the level of acreage planted, selection of cropping patterns, application rates of irrigation water, mix of fresh and saline water, surface water sold to the market, and levels of land retired or idled:

$$[9]\quad \text{Max} \sum_t \left[\frac{1}{(1+r)^t}\right]\{\sum_i [(P_i-H_i)y_{ti}-V_i-K]x_{ti}-\sum_h (w_t^h)W_t^h-(D_t)G_t-(xI_t)K+(xR_t^{'})R+\sum_j (m_t^j)M_t\}$$

where t is year $(t= 1,...,T)$; i is crop $(i=1,...,n)$, and r is real interest rate; x_{ti} is area of crop i in year t; P_i is market price for crop i; H_i is harvest cost per unit of yield; y_{ti} is yield; V_i is per acre non-water variable cost of production; $w^h{}_t$ is total water use by supply source h at price $W^h{}_t$. D_t is drainage volume, and G_t is per unit cost of drainage disposal. K is per acre annual irrigation capital cost, xR'_t is acreage retired for salinity control and R_t is the per acre compensation. Variable $m^j{}_t$ is surface water of supply type j sold in the water market at a price of M_t per acre-foot.

The intertemporal problem in [9] is maximized subject to production function relationships, land and water resource constraints, and initial conditions of certain variables. Several features distinguish this model from the steady state model discussed earlier. First, soil salinity is a state variable in the model, and is included in the production function relationships:

$$y_{ti}=y_{ti}(\sum_j a_{ti}^j,\ C_{ti},\ sL_{ti}) \qquad t=0,...,T; \quad i=1,...,n,$$

$$[10]\quad s_{ti}=s_{ti}(\sum_j a_{ti}^j,\ C_{ti},\ sL_{ti}) \qquad t=0,...,T; \quad i=1,...,n,$$

$$d_{ti}=d_{ti}(\sum_j a_{ti}^j,\ C_{ti},\ sL_{ti}) \qquad t=0,...,T; \quad i=1,...,n.$$

where y_{ti}, s_{ti}, and d_{ti} are per acre yield soil salinity at the end of the irrigation season, and per acre drainage volume. Each of these variables is dependent upon total applied irrigation water per acre $(\Sigma a^j{}_{ti})$, weighted salt concentration of the irrigation water (C_{ti}), and soil salinity following preseason leaching (sL_{ti}). These crop-water production functions were estimated from a multi-year lysimeter experiment conducted under conditions prevailing in the SJV (Dinar et al., 1991c). Technology effects on yield, salinity and drainage are reflected in factor adjustment to base function intercept (Rhoades, 1990)

Water supplies available to the farm include surface (base and supplement) and ground water. Surface water can be used for irrigation during the growing season, for pre-season leaching of soil salts, and for sale in a water market (where permitted). Ground water can also be used for irrigation and leaching. Surface water maybe blended with ground water, and the mix may vary for crop and preseason applications..

Use of saline water may cause accumulation of salts in the soil. The dynamic nature of the problem is driven by equation [11], the equation of motion for soil salinity, based on initial salinity aggregated over acreage base (SI_t), ending soil salinity (s_{ti}), idled acreage (xI_t), and acre base adjusted for land retirement (X_t).

$$[11]\ SI_{t+1}=SI_t\frac{xI_t}{X_t}+\sum_i\left[\frac{s_{ti}x_{ti}}{x_{ti}}\cdot\frac{x_{ti}}{X_t}\right]\qquad t=0,\dots,T.$$

Equation [12] defines the leaching application function, based on initial salinity, soil salinity after leaching (sL_{ti}), weighted salt concentration of leaching water (CL_{ti}), and leaching factor (L).

$$[12]\ \sum_j aL^j_{ti}\geq\frac{L(SI_t-sL_{ti})}{sL_{ti}-CL_{ti}}\qquad t=0,\dots,T;\ i=1,\dots,n.$$

Total drainage produced on farm is the summation over the fields of the drainage produced by the leaching and irrigation applications. Drainage volume produced from crop water applications is computed from Equation [10] above. Drainage volume produced by the leaching activity is calculated as the difference between the amount applied water $(\Sigma_j aL^j_{ti})$ and the root zone water holding capacity (RC-WP), where RC is field capacity and WP is wilting point.

Additional technical and balance equations, upper and lower bounds on certain variables, and initial conditions are included in (Dinar et al., 1991a).

The model is applied to a representative region in the westside SJV over a planning horizon of 15 years. Representative cropping patterns include wheat, sorghum, and wheatgrass, based on availability of production functions from field lysimeter tests. Efforts are underway to estimate yield, soil salinity and drainage (quantity and quality) for a set of major crops on the west side of the SJV. Representative salinity concentrations for surface and ground water are .7 and 2.0 EC, respectively, with an initial soil salinity of 1.5 EC. The application assumes a gravity irrigation system (1/4 mile run), using variable and capital irrigation cost data from CH2M HILL (1989). At this time, irrigation technology choice is exogenous.[4] Prices, technical coefficients and assumptions used for this model can be found in Dinar et al., 1991a.

Several policies and scenarios have been simulated. We include in this presentation only two policy tools (1) water quotas, and (2) drainage disposal permits. The base case is represented by a full water quota of 1500 AF and unrestricted drainage quantity. Impacts of reductions of surface water quota and drainage permits on net present value of income, resulted

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[4]Development is underway on extending the model to incorporate endogenously the adoption of alternative irrigation technologies.

annual optimal drainage volumes, and initial soil salinity are presented in Figure 8 to 11.

Net regional income is plotted against surface water quota and drainage permit levels (Figures 8 and 9). Effects on regional income of drainage permits are modest (reduction of 2% to 33% when permits decrease from 78% to 22% of the base case value) relatively to the surface water quota (reduction of 1% to 50% when quotas decrease from 17% to 40% of the base case value).

Soil salinity and drainage volumes values over time as affected by levels of drainage permits and surface water quotas are ploted in figures 10 and 11. Use of water quota results in relatively lower levels of of soil salinity at the steady state value (around 3 EC), compared to the use of drainage permits (3.05-3.11 EC). However, with drainage permits the steady state value is reached quicker (2-3 years) than with surface water quota (5-7 years). Drainage volumes are reduced by both policy instruments. The optimal path of drainage volumes converges quicker to the steady state values in the case of drainage permits (2 years) compared to surface water quota (2-7 years). The drainage permit becomes an effective constraint after the second year in all levels of drainage permit use.

Annual optimal values for land use and rate of ground water use are provided in Tables 4 and 5. The overall result is under the conditions analyzed here, a steady state solution is achieved relatively early, between one to five years, depending on the variable and the policy instrument used. With quotas on surface water (Table 4), farmers tend to reduce the land used, utilize the surface quota and amend it by ground water pumping. As surface quota decreases, cultivated land is decreased, and ratio of ground water in the applied water mix is increased. Over time, this ratio decreases until reaches the steady state value. For the case of drainage permits (Table 5), the cultivated land does not change, total applied water is reduced as permit levels decrease. Over time there is a slight increase in water application rates, with an increase in the share of ground water. There is also a shift over time to rotations with more wheat. This is more significant as permit level decrease (not presented).

Framework for regional cooperation with irrigation externality problems

The physical-economic models discussed in the previous section provide economically feasible solutions to irrigation externality and non-point pollution problems. However, the suggested policies and solutions may not be acceptable by the parties involved because (1) not all parties were included in the modeling framework, and (2) considerations other than profit maximization are included in the objective functions of some parties.

Here we consider the problem of regional cooperation in water management from a game theory perspective. In contrast to market situations with large number of participants, the game situation in the SJV involves a relatively small number of producers. Producers are organized into water districts, with a board and a water district manager. The district manager has the power to set water rates and water use practices for the district with the acquiescence of the board. An enforcement body exists as well, namely the California Water Resource Control Board.

The current setting has the nature of a noncooperative game. In the noncooperative case, given market prices, participants need to obtain information about preferences of others and need only to choose their own actions based on their own preferences given the actions of others.

Traditional economic solutions for externality problems include use of Pigouvian taxes and Coasian bargaining. Pigouvian tax (Baumol and Oates, 1989) sets the level of the externality at the Pareto optimal level with respect to a noncooperative, rather than a cooperative solution. Coasian bargaining solutions to achieve Pareto optimality, preceded by a required definition of property rights, will also fail to achieve efficient agreement (Samuelson, 1985).

Game theory has previously been applied (e.g., Rinaldi et al., 1979) to externalities and public goods separately, whereas here we apply game theory for a combination of externalities and public goods. A regional cooperative system for water quality/quantity control and improvement has the nature of a public good in that it would provide benefits jointly to producers and consumers (Figure 12) who would value such improvements differently, and require to determine the method for its finance.

Improving water quality for recreation and other instream values would impose costs on agricultural producers, not voluntarily accepted unless offset by benefits of economies of scale and cost sharing schemes. The literature on cost allocation has viewed such situations as cooperative games (Young, 1985; Loehman and Winston, 1971). Recent research in public goods also concerns the free rider problem associated with obtaining demand information by using a demand revealing mechanism to induce truthful behavior as the best strategy. Such schemes generally do not satisfy Pareto optimality in that there will be a budget surplus resulting from the “truth tax”. To avoid this, we assume that a regional manager has access to information on preferences of the players.

We will follow here the framework in Loehman and Dinar (1991) that suggests the use of game theory concepts and a mechanism design approach to the regional externality problem caused by irrigated agriculture. Under this approach, a desired social outcome represents a Pareto optimal cooperative solution which is acceptable to all players and therefore can be sustained as an equilibrium outcome. In this application game players include upslope and downslope producers, consumers of recreation, and a regional manager whose role is to propose and enforce rules of the game. Acceptability of cooperative solutions can be determined for sets of political weights assigned to the parties involved. The mechanism for such game is displayed in Figure 13.

There are three situations in the regional externality problem: (1) the status quo, (2) the noncooperative, and (3) the cooperative solutions (Figure 14). Technical efficiency is represented by a production frontier for agricultural production (F) and environmental amenities (Q). Private technology is applied and operated by the individual producers on their own fields (irrigation systems). Each production technology has an associated frontier, and the noncooperative frontier represents the envelope of the intersections of the frontiers for each private technology. Due to externalities, this frontier maybe convex in the noncooperative case. Cooperative technologies are implemented at the regional level. They may include a regional water treatment facility, regional storage, reuse,drainage systems, and extension and information systems. If cooperative technologies are added to the existing private technologies, the resulting frontier can lie outside the existing frontier for certain combinations of efficiency-quality. The status quo point corresponds to a maximization of agricultural profit with private water use technologies.

Given a set of political weights, with known payoff functions and production relationships, the regional manager than computes taxes and cost shares for both the noncooperative and cooperative solutions by solving the joint maximum problems described below. The noncooperative solution will be achieved through applying Pigouvian taxes determined by the manager. This same set of weights will also be used by the manager to compute the consumers' share of the joint regional costs.

Thus, a higher weight implies more consideration in joint cost. By making cost shares in the cooperative solution equal to political weights used to compute noncooperative and cooperative solutions, consumers will evaluate benefits of environmental quality improvements to the costs of implementing them through the process.

If externalities are severe enough, consumers damaged by externalities may lobby for improved quality standards corresponding to production along the noncooperative frontier. The extent of consumers' success to achieve their objective is affected also by the political power of the producers.

The producers then choose between the pair of noncooperative solutions according to the highest payoff values. Producers can also reacquire that the game continue and political weights be revised if neither the noncooperative nor the cooperative solutions is attractive relative to the status quo.

If such a process stops at a cooperative solution, the equilibrium must be an acceptable cooperative solution such as (1) each player is better of than at the status quo, (2) each player is better off than in the noncooperative solution with the same political weights; and (3) joint costs of cooperation are covered by players.

Game players

Producers payoffs are naturally defined in terms of profits. Following recent environmental literature, the concept of Equivalent Variation (EV) is used to represent consumer preferences for environmental quality in monetary terms reflecting expenditure changes due to changes in environmental quality (health, recreation). Thus, payoffs for consumers and producers will be comparable.

<u>The Producers:</u>

Upslope and downslope producers (u and d, respectively) are characterized with a separate (per acre) production function of water (W) and technology (τ). Each producer has a limited area of land (A). Profits are calculated as revenue from agricultural production less charges for water use (v), taxes on water and land used $(t_\omega^u, t_\ell^u$, respectively), and annual fixed costs for water technologies $c(\tau^u)$.

The upslope producer's yield (Y^u) is related directly to his choices of water and technology. The downslope producer's yield is related to his choices of water and technology, as well as drainage caused by water use of the upslope producer. Both producers maximize profits by choosing optimal levels of cultivated acres, applied water per acre, and the water use technology.

The individual optimization problems for upslope [13] and downslope [14] producers under the status quo case are:

$$[13] \qquad \max_{\tau^u,\, W^u, A^u} \; [p_f Y^u - (v + t_\omega^u) W^u - c(\tau^u) - t_\ell^u] A^u$$

s.t. $A^u \leq \overline{A}^u$

$Y^u = Y^u(W^u; \tau u)$

and:

[14] $\underset{\tau^d,\, W^d, A^d}{\text{Max}} \quad [p_f Y^d - (v + t^d_\omega)W^d - c(\tau^u) - t^d_\ell]A^d$

s.t. $A^d \leq \overline{A}^d$

$Y^d = Y^d(W^d, kW^u A^u; \tau d)$.

Levels of profit given by the optimal solution in the status quo is π^i_0, for producer i.

Both producers generate pollution represented by the concentration of pollutants and the volume of drainage water (This may be a vector were several pollutants are considered). For the case of the up slope producer, pollution depends on land and water use decisions and technologies employed

[15] $S^u = \delta^u(\tau^u)W^u A^u$.

In the case of the downslope producer, pollution discharge depends on drainage from the upslope producer

[16] $S^d = \delta^d(\tau^d)(W^d A^d + kW^u A^d)$.

The total pollution from the region (S) is the sum of the upslope and downslope discharges.

[17] $S = S^u + S^d$

The Consumers:

Preferences of consumers are represented by a utility function. As utility is not defined in dollar units, however, it is not directly comparable to producer profits. Using the Equivalent Variation measure provides a dollar measure of welfare which results in a ranking of outcomes similar to that of a utility-based criteria.

The expenditure function is defined from the indirect utility function:

[18] $\overline{U} = \overline{U}(M, S, p_f, p_h, p_r, p_z)$

where M is initial income, S is the regional drainage discharge, and p. denotes respectively the price of food, health, recreation and other goods. Improvement of drainage quality may improve consumer welfare. The amount of money which is equivalent to a change in the pollution level from the status quo S^O to S' ($S^O > S'$), satisfies the following relationship:

[19] $\overline{U}(M+EV, S^o, p_f, p_h, p_r, p_z) = \overline{U}(M, S', p_f, p_h, p_r, p_z)$.

The EV is a function of drainage water quality in terms of the change in expenditures required to purchase food, health and recreation, relative to the base condition S^o. For simplicity we assume that food prices are not affected by level of production.

In the analysis below, an equivalent variation function, $EV(S;S^o)$, will be used to denote consumer welfare as a function of improved drainage quality. (Note that $\partial EV/\partial S<0$, i.e., as the pollution decreases, the equivalent variation increases.)

The noncooperative Nash Equilibrium and Pigouvian Taxes

The noncooperative Nash Equilibrium (NC) is a game solution in which each player chooses the strategy which maximizes that player's payoffs, given that the strategies of other players are fixed corresponding the noncooperative solution. This solution lies along the production frontier corresponding to choices made by producers among private technologies. A noncooperative solution achieves a tradeoff between drainage water quality and agricultural production and is indicated by the slope of the production frontier. Each solution can be related to a given set of political weights of the players.

The frontier is found by maximizing a weighted sum of payoff functions for game players with varying weights summing to one (Takayama, 1974). The joint welfare optimization problem is:

$$[20]\quad JW(\alpha;NC)= \max_{\substack{\tau^u,\, W^u, A^u,\\ \tau^d,\, W^d, A^d}} \alpha_c EV(S;S^o)+\alpha_u[p_f Y^u - vW^u - c(\tau^u)]A^u+\alpha_d[p_f Y^d - vW^d - c(\tau^d)]A^d$$

$$\text{s.t.}\quad \begin{aligned} & A^u \le \overline{A}^u \\ & A^d \le \overline{A}^d \\ & A^u W^u + A^d W^d \le \overline{\overline{W}} \\ & Y^u = Y^u(W^u;\tau^u) \\ & Y^d = Y^d(W^d, kW^u A^u;\tau^d) \\ & S^u = \delta^u(\tau^u) W^u A^u \\ & S^d = \delta^d(\tau^d)(W^d A^d + kW^u A^d) \\ & S = S^u + S^d \end{aligned}$$

Here, the weighted sum of producer and consumer payoffs is maximized over private technologies, irrigated acres, and water use. Constraints are the same as for the individual maximization problems in [13] and [14], except that there is a regional water constraint $\overline{\overline{W}}$. For the noncooperative solution corresponding to political weights α, the optimal pollution level is denoted by $S(\alpha;NC)$. (For the status quo, the weight on the consumer is zero.)

The noncooperative solution is achieved as a Nash equilibrium by producer profit maximization in response to appropriate taxes (e.g. Pigouvian tax) set by a regional authority. Pigouvian taxes to achieve a given noncooperative equilibrium are derived from first order conditions for the noncooperative joint maximum problem in [20]. Since the pollution is a non-point problem and pollution is determined by land and water use, taxes on pollution are equivalent to taxes on land and water (assuming knowledge of the physical relationships) which are preferred due to reduced information and enforcement costs.

Solving the frost order condition for marginal profit, the optimal taxes on water use and land for the upslope and downslope producers, respectively, are represented by the right handside of the following expressions:

$$[21]\ \frac{\partial \pi^u}{\partial W^u} = \frac{\mu}{\alpha_u}A^d - \frac{\alpha_c}{\alpha_u}\frac{\partial EV}{\partial S}(\delta^u + \delta^d k)A^u - \frac{\alpha_d}{\alpha_u}\frac{\partial \pi^d}{\partial \pi^u}$$

$$[22]\ \frac{\partial \pi^d}{\partial W^d} = \frac{\mu}{\alpha_d}A^d - \frac{\alpha_c}{\alpha_d}\frac{\partial EV}{\partial S}\delta^d A^d$$

$$[23]\ \frac{\partial \pi^u}{\partial A^u} = \frac{\lambda_u}{\alpha_u} - \frac{\alpha_c}{\alpha_u}\frac{\partial EV}{\partial S}(\delta^u + \delta^d k)W^u - \frac{\alpha_d}{\alpha_u}\frac{\partial \pi^d}{\partial A^u}$$

$$[24]\ \frac{\partial \pi^d}{\partial A^d} = \frac{\lambda_d}{\alpha_d} - \frac{\alpha_c}{\alpha_d}\frac{\partial EV}{\partial S}\delta^d W^d.$$

The shadow price for the regional water constraint is denoted by μ, and $\lambda_u\ \lambda_d$ represent land opportunity values for the upslope and downslope producers, respectively.

Note that the taxes $t^i_\omega(\alpha;NC)$; $t^i_\ell(\alpha;NC)$ for each producer i are related to political weights α. Optimal taxes on water and land use for the upslope producer should be higher than for the downslope producer for equal weights and area planted, because upslope producers cause external costs for both down slope producers and consumers. In the optimal solution, water use is reduced relative to the status quo case where no taxes are imposed and the marginal profit equals zero (Figure 15). By the same token, less land will be irrigated when taxed, relative to the status quo.

Producers' profit for the noncooperative solution are obtained by subtracting taxes from profits in the joint maximum

$$[25]\ \pi^i(\alpha;NC) = [p_f Y^i - vW^i - c(\tau^i)]A^i - t^i_\omega(\alpha;NC)W^i A^i - t^i_\ell(\alpha;NC)A^i \quad ,\forall i=1,2.$$

Cooperative solution

In the cooperative case, regional technologies for drainage reduction and treatment are available in addition to private technologies. In the cooperative case, reduced pollution levels may be achieved at lower cost under regional facilities due to economies of scale.

The joint welfare problem solved for the cooperative and noncooperative cases are similar except that for an acceptable solution, the joint costs in the cooperative case should not exceed the difference between the weighted sum of payoffs in the cooperative and noncooperative cases. The cooperative solution is found without reference to the method of cost allocation. The optimization problem for the cooperative production frontier is:

$$[26]\ JW(\alpha;CS)=\max_{\substack{\tau^R,\\ \tau^u,\ W^u, A^u,\\ \tau^d,\ W^d, A^d}} \alpha_c EV(S;S^o)+\alpha_u[p_f Y^u - v'(\tau^R)W^u - c(\tau^u)]A^u + \alpha_d[p_f Y^d - v'(\tau^R)W^d - c(\tau^d)]A^d - JC(S,\ W^R;\tau^R)$$

s.t.

$$A^u \le \overline{A}^u$$
$$A^d \le \overline{A}^d$$
$$A^u W^u + A^d W^d = W^R \le \overline{W}\, r(\tau^R)$$
$$Y^u = Y^u(W^u;\tau u)$$
$$Y^d = Y^d(W^d,\ kW^u A^u;\tau d)$$
$$S^u = \delta^u(\tau^u)W^u A^u$$
$$S^d = \delta^d(\tau^d)(W^d A^d + kW^u A^d)$$
$$S = S^u + S^d.$$

Regional technologies are denoted by τ^R, regional cost of treating and reusing drainage water is $JC(S\ ,W^R;\tau^R)$, where W^R is total volume of water used in the region. Pollution S is related to the weights in the cooperative solution and is denoted by $S(\alpha;CS)$. Charges for water use (v') may be smaller than the charge for water in the noncooperative case (v), because of reuse.

Comparison of the cooperative (CS) and the NC joint maximum problems shows that a cooperative solution will result in a higher value for the objective function JW since private technologies are feasible for both the cooperative and noncooperative problems.

As in the noncooperative solution, imposing Pigouvian taxes on land and water use makes private water and land use decisions consistent with the joint welfare maximum. In addition to externality effects, the tax now includes marginal (variable) cost for the joint facility. Because of economies of scale, revenue from these taxes will not cover costs of the joint facility. Therefore, agreement to participate in the cooperative solution requires that producers pay a share of the joint cost (including fixed cost) of the regional facility.

Producer's profit in the cooperative solution, after taxes and cost shares are:

$$[27]\ \pi^i(\alpha;CS) = [p_f Y^i - v'(\tau^i)W^i - c(\tau^i)]A^i - t^i_\omega(\alpha;CS)W^iA^i - t^i_\ell(\alpha;CS)A^i - \alpha_i[JC(S, W^R;\tau^R) - TR] \quad ,\forall\ i=1,2$$

where TR denotes total tax revenues collected in the region.

Acceptable solutions

For a solution of [26] to be "acceptable", requires that all parties prefer a set of payoffs to both the noncooperative solution and the status quo, so that such a solution could be achieved voluntarily. For producers, two conditions must apply. First, the payoff in the cooperative case must be greater then in the noncooperative case

$$[28]\ \pi^i(\alpha;CS) - \pi^i(\alpha;NC) \geq 0,$$

and payoff in the cooperative case should exceed profits in the status quo case

$$[29]\ \pi^i(\alpha;CS) - \pi^i_o \geq 0.$$

If [29] holds, than enforcement cost can be minimized.

As mentioned before, the technology choice set for the noncooperative problem is contained in that for the cooperative problem. Therefore, profits will be greater in the cooperative solution than in the noncooperative solution, if (1) the joint cost share is less than the tax cost in the noncooperative solution, (2) private technologies are less expensive in the cooperative case, and (3) output is not reduced in the cooperative case.

Consumers are better off in the noncooperative case compared to the status quo since pollution is reduced. Pollution is at least the same in the cooperative case compared to the noncooperative case. However, since consumers do not have to pay in the noncooperative case, consumers are only better off in cooperative solution when:

$$[30]\ EV[S(\alpha;CS);S^o] - EV[S(\alpha;NC);S^o] \geq \alpha_c[JC-TR].$$

That is, water quality in the cooperative solution must be sufficiently higher than in the noncooperative solution to offset the cost share paid by consumers in the cooperative case.

Whether equations [28]-[30] hold will depend on the political weights, nature of the physical relationships and available technologies.

Application

The approach described above was applied to conditions in the SJV using a simplified example. Upslope and downslope producers grow the same crop with two irrigation technology options. Upslope drainage affects downslope water quality, and total drainage produced in the agricultural process pollutes a receiving water body which serves as a recreation source. The drainage water can be treated in a regional plant before discharged to the water body. Physical relationships, agricultural production costs, treatment cost, and consumer benefits are estimated and incorporated in the application.

Results are presented in Table 6 and Figure 16. Table 6 gives payoff values for the status quo, the noncooperative and cooperative solutions, for various consumers' weights. Figure 16 shows the corresponding production frontiers for noncooperative and cooperative cases. The nonconvexity of the noncooperative frontier reflects both externalities as well as indivisibilities of private technology choices.

For the cooperative solution with political weight of .33 for the consumers, area farmed and water applied is reduced, less drainage is produced because of increased irrigation efficiency, and the amount of drainage treated increases. As political weight assigned to consumers is increased, pollution is reduced although the consumer bears a larger share of cost obtained but also the consumer has a larger cost share. Producers reduce cultivated area to meat the quality constraint. For low pollution levels, drainage is reduced and producers pay a smaller share of the joint facility cost.
The consumer wieght of .40 produces an acceptable cooperative solution for the cost sharing method of shares equal to political weights.

Discussion, and future research needs

This paper deals with several problems. First, it demonstrates the complexity of physical relationships that are associated with agricultural irrigation pollution. Second, it suggests ways to overcome these complexities and still provide meaningful information to policy makers. Third, it argues that, given the case of nonpoint source pollution and externalities, cooperation between the parties involved and voluntary solutions may provide under certain conditions an easier way to achieve socially preferred policies.

During the course of our research on irrigated agriculture and environmental pollution in the SJV, we made several compromises,however we gained much experience and passed several junctions. We feel that we can now make several generalizations based on our research results, and would be happy to open it for discussion.

Appropriate modeling of the interacting system is essential for providing relevant impact and policy analysis. How do we do that? The instant answer is an interdisciplinary work including scientists that are familiar with the technical aspects of physical relationships. This means that we, the economists, need to collaborate with hydrologists, soil and plant science experts and environmentalists. We must suggest them what are the important variables that we need, and urge them to provide us with information and data that can be implemented by us in our models. Having appropriate data set is important for our analysis because although theoretical relationships may exist, their implementation for policy analysis may not be relevant.

Another important feature of our analysis is aggregation. Since our capacity to analyze properly real world economic and/or physical relationships is limited, we are facing a problem. Aggregation, if done properly, may reduce the burden. Aggregation may take place either over the parties involved in the problems to be solved, or over variables affecting the system.

Dynamic versus steady state approaches to model physical relationships as well as economic behavior have much been discussed in the past. Unfortunately, our data did not allow us to model the exact same problem under both dynamic and steady state approaches. It is clear the the dynamic approach provides a better and probably a more realistic description of the behavior of key variables. However, it also introduces an addition burden to the modeler. In the example introduced in this paper, for the initial condition used (soils salinity and water qualities), the additional information gained using the dynamic approach was very marginal since a convergence to steady state was reached very early. Application of the model under more extreme conditions will result in a different optimal behavior compared to a steady state model.

A non-relevant set of policy variables and instruments chosen by the policy maker, may misguide the analysis. For example, in the specific case of drainage water, there is a reciprocal relationship between discharged volume and the concentration of pollutants. Dealing with one only may mislead the policy maker.

Finally, the question whether acceptable cooperative versus the status quo or noncooperative solutions in a real world. For implementation of the mechanism suggested here, it is assumed that institutions already exist for data collection, computation of taxes, and dissemination of information. Even with potential gains, actual acceptance of cooperative solution is a remaining question. Further behavioral work should be undertaken to determine whether a cooperative game process such as that proposed here would result in an actually acceptable cooperative solution.

Table 1:

Acres, drainage conditions, and potential yield levels by farm (steady state model).

Farm No.	Potential Cultivated land (ha)	Irrigation to Drainage Ratio (fraction)	Potential yield level (ton/ha)			
			Alfalfa	Cotton	Tomato	Wheat
1	1875	.5	21.25	1.6	78	3.5
2	1775	.8	17.50	1.6	72	3.0
3	400	.2	21.25	1.6	85	3.5

Table 2:

Regional income, acres farmed, applied surface and ground water, and drainage quantity and quality, and collected taxes, as affected by policy measures (steady state model)

Policy and policy var. value	Regional income[a] ($$10^6$)	Acres farmed Alf. (ha)	Acres farmed Cot. (ha)	Acres farmed Tom. (ha)	Applied water Surface (10^6 ha cm)	Applied water Ground (10^6 ha cm)	Drainage water Quantity (10^6 ha cm)	Drainage water Quality (EC)	Collected taxes ($$10^6$)
No regulation	1148.4	200	2174	1150	355.5	53.7	63.0	8.3	0
Water price (flat)									
$3/ha cm	970.7	200	2174	1150	355.5	53.2	62.8	8.6	177.7
$6/ha cm	300.4	200	1425	1150	101.8	198.9	33.2	14.7	356.2
Drainage fee (flat)									
$10/ha cm	763.1	200	1425	1150	255.3	23.1	24.6	21.5	246.8
$40/ha cm	277.5	200	1106	937	204.7	2.5	10.4	56.7	417.1

a Not including redistribution of taxes

Table 3:

Irrigation technology selected, area farmed, Applied surface and ground water by farm and policy measure (steady state model).

Policy	Farm 1	Farm 2	Farm 3
	CUC of irrigation technology used to Irrigate the main crop (cotton)		
No regulation	75	75	75
Water price-Flat \$3/ha cm	75	75	75
Water price-Flat \$6/ha cm	75	75	75
Drainage tax-flat \$10/ ha cm	75	75	75
Drainage tax-flat \$40/ha cm	80	87	75
		Area farmed (ha)	
No regulation	1875	1250	400
Water price-Flat \$3/ha cm	1875	1250	400
Water price-Flat \$6/ha cm	1875	500	400
Drainage tax-flat \$10/ ha cm	1875	500	400
Drainage tax-flat \$40/ha cm	1844	0	400
		Applied surface water (ha cm)	
No regulation	168750	150750	36000
Water price-Flat \$3/ha cm	168750	150750	36000
Water price-Flat \$6/ha cm	101771	0	0
Drainage tax-flat \$10/ha cm	168750	49177	36000
Drainage tax-flat \$40/ha cm	168750	0	36000
		Applied ground water (ha cm)	
No regulation	44137	0	9594
Water price-Flat \$3/ha cm	43722	0	9459
Water price-Flat \$6/ha cm	99999	51086	46434
Drainage tax-flat \$10/ ha cm	16000	0	7084
Drainage tax-flat \$40/ha cm	0	0	2497

Table 4:

Optimal annual values for land and water use in the case of reduced surface water quota

	Cropland use				Total water applied			
Year	------Surface water quota------				------Surface water quota ------			
	1500	1250	750	500	1500	1250	750	500
1	500	500	427	281	1816	1816	1443	957
2	500	500	322	234	2108	2039	1258	872
3	500	500	313	220	2117	2037	1243	847
4	500	500	310	213	2118	2037	1238	836
5	500	500	309	210	2118	2037	1236	831
6	500	500	309	208	2118	2037	1236	827
7	500	500	309	207	2118	2037	1236	826
8	500	500	309	207	2118	2037	1236	825
9	500	500	309	207	2118	2037	1236	825
10	500	500	309	207	2118	2037	1236	825
11	500	500	309	207	2118	2037	1236	825
12	500	500	309	207	2118	2037	1236	825
13	500	500	309	207	2118	2037	1236	825
14	500	500	309	**207**	2118	2037	1236	825
15	500	500	309	207	2118	2037	1236	825

Table 5:
Optimal annual values for land and water use in the case of drainage permits

	Cropland use				Total water applied			
Year	------Drainage permit ------				------Drainage permit ------			
	450	350	250	100	450	350	250	100
1	500	500	500	500	1816	1816	1816	1791
2	500	500	500	500	2108	2025	1924	1757
3	500	500	500	500	2117	2026	1925	1753
4	500	500	500	500	2118	2027	1925	1752
5	500	500	500	500	2118	2027	1925	1752
6	500	500	500	500	2118	2027	1925	1752
7	500	500	500	500	2118	2027	1925	1752
8	500	500	500	500	2118	2027	1925	1752
9	500	500	500	500	2118	2027	1925	1752
10	500	500	500	500	2118	2027	1925	1752
11	500	500	500	500	2118	2027	1925	1752
12	500	500	500	500	2118	2027	1925	1752
13	500	500	500	500	2118	2027	1925	1752
14	500	500	500	500	2118	2027	1925	1752
15	500	500	500	500	2118	2027	1925	1752

Table 6:
Payoff results ($000)

Status quo

	Consumers benefits	Producer 1 payoff	Producer 2 payoff	Pollution ppb Se
	248	825	516	31.91

Noncooperative solutions

Weight on consumers	Consumers benefits	Producer 1 payoff	Producer 2 payoff	Pollution ppb Se
.60	284	657	422	22.14
.50	279	712	436	23.54
.42	271	713	440	25.64
.40	271	711	440	25.64
.33	268	707	443	26.44

Cooperative solution

Weight on consumers	Consumers benefits	Producer 1 payoff	Producer 2 payoff	Pollution ppb Se
.60	265	799	495	14.43
.50	286	804	533	15.43
.42	271	837	542	15.64
.40	272	835	541	15.64
.33	252	834	541	22.55

Benefit/Loss of cooperative Solutions as related to weights

Weight on consumer	Consumers	Producer 1	Producer 2
	Compared to NC	Compared to SQ	
.60	-19	-26	-21
.50	+7	-21	+17
.42	0	+12	+26
.40	+1	+10	+25
.33	-16	+9	+25

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Figure legend: